South Dakota State University
Open PRAIRIE: Open Public Research Access Institutional Repository and Information Exchange

Theses and Dissertations

2017

Biotic Integrity in the Northwestern Great Plains and Mechanisms Regulating Stream Condition in South Dakota

Chad Kaiser

BIOTIC INTEGRITY IN THE NORTHWESTERN GREATPLAINS AND MECHANISMS REGULATING STREAM CONDITION IN SOUTH DAKOTA

BY

CHAD KAISER

A thesis submitted in partial fulfillment of the requirements for the

Master of Science

Major in Wildlife and Fisheries Sciences

Specialization in Fisheries

South Dakota State University

2017

BIOTIC INTEGRITY IN THE NORTHWESTERN GREAT PLAINS AND MECHANISMS REGULATING STREAM CONDITION IN SOUTH DAKOTA

This Thesis is approved as a creditable and independent investigation by a candidate for the Master of Science degree in Wildlife and Fisheries Sciences and is acceptable for meeting the thesis requirements for this degree. Acceptance of this does not imply that the conclusions reached by the candidate are necessarily the conclusions of the major department.

Katie N. Bertrand, Ph.D. Date
Major Advisor

Michele Dudash, Ph.D. Date
Head, Department of Natural Resource Management

Kinchel Doerner, Ph.D. Date
Dean, Graduate School

ACKNOWLEDGEMENTS

I would like to thank my major advisor Dr. Katie Bertrand. My life has been defined by opportunities, as an undergraduate advisor Dr. Bertrand presented me with my first opportunity to work in fisheries. This single opportunity and her constant support and trust has had a profound impact in molding my career in fisheries and for that I will be forever grateful. You have been a mentor to me in so many ways, thank you.

I would also like to thank Dr. Nels Troelstrup for his technical guidance and support throughout this project. I would like to thank my committee members Dr. Steven Chipps and Dr. Robin Arends. Next, I would like to thank Dr. Brian Graeb for consistently scratching your chin and asking your students, "What do you think?" I will carry this with me for the rest of my life and try to remember to keep asking questions. Terri Symens, Kate Tvedt, Di Drake, and Dawn Van Ballegooyen you all are/were the back bone of this department, thank you so much for your help along the way. Also, I would like to thank the South Dakota Department of Environment and Natural Resources.

I need to thank Lyntausha Kuehl, without your dedication this project would not have succeeded. You have provided a much appreciated discipline to my seemingly chaotic daily operation. And to Aaron, the GIS wizard behind the curtain, thank you for your contribution to this project.

To the technicians I have had the honor of sampling with, my success is because of you. Logan Gutzmer, thank you for all of your hard work. You made my life so much easier by being so willing to work hard. I will never forget your ability to sleep anywhere, including with me. Zach Jesse, thank you for knowing every road in South Dakota, having family or ex-girlfriends willing to put us up, and for being the best copilot I could ask for. Scott Sabo thank you for teaching us a little dadgummit and for being the best animal wrangler I have had the pleasure of meeting. For the record the biscuit is an inferior form of breakfast sandwich enclosure, science. Jordon Redmond the wader wrestling champion of the west, thank you, and thank you for not letting that guy kill us that one time. Finally Calvin Rezac, thank you. Thank you for stepping in front of that bee and taking one for the team. You have been and absolute pleasure to work with and it has been so rewarding to watch you develop professionally. I expect big things from you. I have said that you learn the most from terrible bosses and truly great bosses, I hope you all have learned a lot.

I also need to thank my fellow graduate students, so many of you have helped me the list could include almost anyone who has been at SDSU in past few years. I only hope that I have given back as much as you all have given me. Specifically, my lab group John Lorenzen, Will Radigan, Chance Kirkeeg, Seth Fopma, Eli Felts (you have been a constant professional and someone I admire), Jason Breggemann, Nick Kludt, and Katie Schlafke. Finally from the lab group Stephen Jones my west river brother thank you. Matt Wagner, I cannot thank you enough for being my friend, you have been one of the best things that happened to me in grad school. Thank you to Jacob Krause, for teaching me how to IBI before I knew I would need to know how to IBI, without you I may have gotten lost. Finally David Schumann, you have given so much to me, listening to you work through and think about fish ecology has raised the bar of what I expect from myself. Thank you for your insights, advice, and for listening.

Finally to my family: Christine, Molly, Paige, and Claire, without you this would have been completed much sooner. Christine you have been an ever present force driving me to be my best. Thank you so much for your unwavering support. Molly, Paige, and Claire thank you for keeping me focused on what is really important in life. To my mom, thank you for always being there and for believing in me even when I do not. I would like to thank my brothers, for their support throughout my life. Finally I would like to thank my extended family thank you so much for enduring this ride with us.

Funding for this work was provided by the South Dakota Department of Natural Resource Management and South Dakota State University Department of Natural Resource Management. Additional support was provided to K. Bertrand from the SD Agricultural Experiment Station through Hatch fund 3AH-512.

CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

BIOTIC INTEGRITY IN THE NORTHWESTERN GREAT PLAINS AND MECHANISMS REGULATING STREAM CONDITION IN SOUTH DAKOTA

CHAD KAISER

2017

Anthropogenic disturbance of streams can alter biotic integrity in various ways. Some degradation is easy to classify and monitor, others such as habitat impairment may be less easy to quantify. The Index of Biotic Integrity (IBI) is a unique method of assessing the aquatic health of an ecosystem. Beginning in 2010 the South Dakota Department of Environment and Natural Resources (SD DENR) began implementing biological monitoring on wadeable streams by developing an IBI for the Northern Glaciated Plains ecoregion in eastern SD (Krause et al. 2013). Prior to this survey the condition of the majority of SD's streams was unknown. As the SD DENR expands biological monitoring into the Northwestern Great Plains ecoregion, multimetric indices.

Western South Dakota (SD) streams were lacking a prairie stream water quality assessment. The work presented here in will be an expansion on this previous multi metric index from eastern SD into the Northwestern Great Plains (NWGP) ecoregion of western SD. Chapter one focuses on first developing indices of biotic integrity for fish for the NWGP ecoregion and second identifying regional candidate reference sites by applying statistical distributions defined from field data and multivariate discriminant analysis and ATtILA to validate those candidate reference sites. For the development of the IBI, 65 sites were sampled in the NWGP ecoregion and represent a stratified random sample based on the number of perennial wadeable streams within the smaller Level IV

ecoregion. Metrics were calculated by assessing fish life history characteristics and placed into nine classes. Metrics were then screened and using a sequential series of statistical evaluation. The final IBI consisted of six metrics that will be used to describe the condition of streams in western SD.

The second chapter focuses on the habitat drivers of the IBI. We used the IBI metrics identified for the Northern Glaciated Plains and Northwestern Great Plains ecoregions of South Dakota to represent the attributes of community structure that were most sensitive to anthropogenic disturbance. We then assessed the relationship between habitat variables measured as part of the Environmental Monitoring and Assessment Program (EMAP) protocols and the IBI metrics for each region. These associations move the IBI beyond characterizing stream integrity to identifying factors that could be manipulated to improve or degrade stream integrity. Through these assessments mangers could formulate management plans to improve water quality and subsequently improve IBI scores.

CHAPTER 1. INTRODUCTION

Freshwater comprises only 0.01% of the total volume of the world's water and covers only 0.8% of the Earth's surface (Gleick 1996). However, freshwater contains 40% of all fish diversity and 25% of all vertebrate diversity (Dudgeon et al. 2005). Extinction has reached an unprecedented rate in the Holocene, which some argue should be renamed the Anthropocene (Waters et al. 2016), in acknowledgment of the effects of humans on all other living and non-living things. Thus, maintaining biodiversity may be more critical to the continued provisioning of ecosystem goods and services than ever. Extinction rates are five times higher in freshwater ecosystems (Ricciardi and Rassmussen 1999) than in terrestrial ecosystems (Sala et al. 2000). Maintaining biodiversity increases the buffering capacity of nutrient perturbations and exotic species invasions (Balvanera et al. 2006). Just as increased diversity in crops improves crop resistance, biodiversity in streams enables functional process to persist, increasing the diversity of functions performed by the ecosystem and making resource use more efficient (Chapin 1997).

In 1972, the U.S. Congress enacted the Clean Water Act (CWA), which aimed to "restore and maintain the chemical and physical integrity of the Nation's waters" (Clean Water Act 1972). Initially the Clean Water Act focused on specific contaminants and did not consider the system or community of biotic organisms (Karr 1981). After a veto by President Richard Nixon, the Senate Committee on Environment and Public Works concluded, "chronic adverse biological impact may be a greater problem than the acute results of discharge of raw sewage or large toxic spills" (Karr and Chu 1999). In so doing, the CWA recognized that the nation needed to implement a biological monitoring program.

The Clean Water Act (CWA) adopted total maximum daily load (TMDL) measurements to assess water quality. Though assessments of TMDL provide an accurate measure of those parameters, it is not a time-integrated assessment and may miss non-point source perturbations. With regard to excess fertilizer (nitrogen and phosphorus) inputs, riparian zone degradation, and stream siltation, 42% of the USA's wadeable, perennial streams and rivers were found to be in poor condition (Paulsen et al. 2008). These deviations from the original undisturbed condition affect the resident biotic community, and the community responds with changes in abundance, persistence or extinction of individual species (Heitke et al. 2006). Furthermore, monitoring for water pollution alone neglects underlying geology and anthropogenic perturbations of habitat and flow modifications (Karr 1981).

Karr first identified the opportunities for improving the original sampling methodology of the CWA in 1981. Those challenges were addressed by taking a more holistic approach to water quality and anthropogenic disturbance monitoring by assessing the resident community of the water bodies. In Karr's first IBI (1981) he assigned fish to six classes that encompassed 12 metrics, these metrics were then compared to reference sites (i.e., minimally disturbed sites). The most common methods for identifying reference sites are: 1) best professional judgment, 2) interpretation of historical condition, 3) ambient distribution, and 4) empirical models (Stoddard et al. 2006). A disadvantage of best professional judgment is that the professional could be biased or wrong. For example, best professional judgment could be based on fishing condition, a good fishing stream may not characterize an undisturbed condition, alternatively, a reference site may not be representative of the streams in that region (Whittier et al. 2007). Whittier et al. (2007) indicated that probability based (i.e., empirical models) reference site identification is preferable to best professional judgment for indicator development and biological assessments, specifically in plains regions. The IBI is a unique method of assessing the aquatic health of an ecosystem. In Chapter 2, I provide details of the development of a fish index of biotic integrity for the Northwestern Great Plains Ecoregion of South Dakota.

The work presented in the third chapter will expand on the IBI developed in Chapter 2 and on the fish IBI developed by Krause et al. (2013) for the Northern Glaciated Plains (NGP) Ecoregion to identify habitat variables that could be manipulated to improve or degrade stream integrity in these two ecoregions. This chapter also serves as a framework to guide others in translating IBI scores into actionable management plans. Assessments of biotic integrity are critically important in classifying and monitoring streams in South Dakota. One of the drawbacks of the IBI is that once site scores are calculated those scores make no reference to the disturbance that caused the high or low scores. Index of Biotic Integrity scores are derived based on the fish present at a stream at the time of its assessment. To improve IBI scores at a given site, positive metrics must increase and/or negative metrics must decrease. This is a simple theory, but in reality, the fish must be able to move freely throughout the region and meet their basic life requirements, in order to establish in a “restored” reach. With the completion of the first chapter of this thesis and the IBI completed by Krause et al. (2013) in the NGP, SD now has two fish IBI’s that assess disturbance of about 80% of the state. The objective of the third chapter was to identify and assess the habitat variables from two Level (LV) III ecoregions (NGP and NWGP) in South Dakota that are most influential on fish communities. Environmental Monitoring and Assessment Program (EMAP) protocols, plus additional spatial scale habitat variables. This comparison was conducted by using multivariate statistical modeling programs. Multiple analyses were used to assess the most influential habitat variables affecting fish species distributions in the NGP ecoregion and NWGP ecoregion. The habitat variables more associated with positive metrics would be an environmental variable that potentially has a positive impact on water quality. Alternatively, any negatively charged metric associated with habitat variables could show environmental variables which have degraded enough to negatively affect stream quality. We hypothesized that by assessing the drivers of community structure we should be able to forecast potential fish habitat related stressors that affect assemblage distributions and either prevent degradation to critical fish community habitats or identify actionable habitat features for mitigation.

Literature Cited

Balvanera, P., A. B. Pfisterer, N. Buchmann, J. He, T. Nakashizuka, D. Raffaelli, and B. Schmid. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters 9:1146-1156

Chapin, F.S. 1997. Biotic Control Over the Functioning of Ecosystems. 277 (5325): 500-504

Dudgeon, D. 2005. River rehabilitation for conservation of fish biodiversity in monsoonal Asia. Ecology & Society 10: 15.

Gleick, P.H. 1996. Basic water requirements for human activities: Meeting basic needs. Water International 21: 83-92.

Heitke, J. D., C. L. Pierce, G. T. Gelwicks, G. A. Simmons, and G. L. Siegwarth. 2006. Habitat, land use, and fish assemblage relationships in Iowa streams: Preliminary assessment in an agricultural landscape. Landscape Influences on Stream Habitats and Biological Assemblages 48:287-303.

Karr, J. R. 1981. Assessment of Biotic Integrity Using Fish Communities. Fisheries 6(6): 21-27

Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington D. C.

Paulsen, S. G., A. Mayio, D. V. Peck, J.L. Stoddard, E. Tarquinio, S.M. Holdsworth, J. Van Sickel, L.L. Yuan, C.P. Hawkins, A.T. Herlihy, P.R. Kaufmann, M.T. Barbour, D.P. Larsen, A.R. Olsen. 2008. Condition of stream ecosystems in the US: an overview of the first national assessment. Journal of the north american Benthological society 27(4): 812-821.

Ricciardi, A., and J.B. Rasmussen. 1999. Extinction rates of North American freshwater fauna. Conservation Biology 13(5): 1220-1222.

Sala, O. E., F. S. Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L.F. Huenneke, R.B. Jackson, A. Kinzing, R. Leemas, D.M. Lodge, H.A. Mooney, M. Oesterheld, N.L. Poff, M.T. Sykes, B.H. Walker, M. Walker, and D.H. Wall. 2000 Global biodiversity scenarios for the year 2100. Science 287(5459): 1770-1774.

Stoddard, J. L., D. P. Larsen, C. P. Hawkins, R. K. Johnson, and R. H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications 16(4):1267-1276.

Whittier, T. R., R. M. Hughes, G. A. Lomnicky, and D. V. Peck. 2007a. Fish and amphibian tolerance values and an assemblage tolerance index for streams and rivers in the western USA. Transactions of the American Fisheries Society 136(1):254-271.

Whittier, T. R., and coauthors. 2007b. A structured approach for developing indices of biotic integrity: Three examples from streams and rivers in the western USA. Transactions of the American Fisheries Society 136(3):718-735.

CHAPTER 2: Fish Index of Biotic Integrity for Wadeable Perennial Steams of the Northwestern Great Plains Ecoregion of Western South Dakota

This chapter is being prepared for submission to Ecological Indicators and was co-authored by Katie N. Bertrand, Lyntausha Kuehl, Aaron Suehring, and Nels H. Troelstrup, Jr.

Abstract

Water quality monitoring through biological communities provides a time integrated measure of anthropogenic disturbance. Fish are responsive to disturbance and comprise features of ecosystem structure and function, and fish represent different trophic levels, habitat and reproductive guilds, and varying tolerance levels. All of these features combine to form metrics in indices of biotic integrity. Fish data were used to create an index of biotic integrity (IBI) for the Northwestern Great Plains (NWGP) ecoregion in western South Dakota. Fish were sampled from 65 sites. Sample reaches were stratified by Level IV ecoregion, and the number of sites in each ecoregion was proportional to the number of river kilometers in that ecoregion. Metrics belonged to nine metric classes and each was assessed for responsiveness to anthropogenic disturbance. Optimal metrics were selected through a filtering process using range, signal-to-noise ratios, responsiveness to disturbance, and redundancy tests until there was one metric remaining in each class. The final IBI contained six metrics that best delineated anthropogenic disturbance in the NWGP ecoregion. There was a significant statistical difference between scores of least disturbed sites and most disturbed sites ($F_{1,21}$ = 27.21, P < 0.00) with least disturbed sites scoring 50% higher on average than most disturbed sites (mean ± SE; $\bar{x}$ = 62.22 ± 5.06; $\bar{x}$ = 31.36 ± 2.77). The NWGP IBI provides a tool for monitoring water quality in western South Dakota and a baseline of biotic condition in this ecoregion.

Introduction

Maintaining healthy ecosystems is paramount in an era of exponential human population growth and resource use (Gleick 1996). All human populations rely on ecosystem goods and services. The goods we use from river and stream ecosystems, include fish for food and fresh water for consumption (Balvanera et al 2006). As goods exploitation increases, ecosystem integrity generally decreases (Berka et al. 2001; Dudgeon 1992; Gleick and Palaniappan, 2010). River and stream ecosystems also provide services, including drinking water filtration, fish and wildlife habitat, and aesthetics (Loomis 2000). Other services that are less readily observed include nutrient storage, energy conversion, and the regulation of seasonal discharge (De Groot et al 2002).

James Karr (1981) first proposed the idea of using fishes to monitor biological integrity. Frey (1977) and Karr and Dudley (1981) described biotic integrity as “the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of that region”. This biotic integrity is based not only on “pre-Columbian” composition but is a reference to function or services comparable to undisturbed conditions (Hughes et al. 1998). Since its design in 1981 the Index of biotic Integrity (IBI) has spread from the Midwest of the United States and into many other regions, countries and continents (Ruaro 2012). The original IBI contained 12 metrics, or biological attributes, and has evolved to over 200 starting metrics before the screening process, where each metric is a hypothesis of how that biological attribute will respond to human influence (Karr 1981, Bramblett et al. 2005; Krause et al. 2013; Whittier et al. 2007). To be effective each metric must be sensitive to anthropogenic perturbations but unresponsive to natural gradients (Bramblett etal. 2005). Fish represent an ideal candidate as a biologic indicator because they require a minimal amount of gear and time to both sample and identify and are present in all but the most degraded waters (Fausch et al.

1984). Fish are responsive to disturbance and comprise features of ecosystem structure and function, and fish represent different trophic levels, habitat and reproductive guilds, and varying tolerance levels (Bramblett et al. 2005; Fausch et al. 1984; Whittier et al. 2007).

Stream fish assemblages are ideal candidates because the assemblage encompasses the synergistic effects of water quality degradation (e.g. specific contaminants), sublethal effects (e.g. DO and siltation), and habitat degradation, while being less vulnerable to annual variation and easily captured and identified (Hughes et al. 1998; Karr 1981; Karr and Chu 1999). The IBI is made more robust by including multiple metrics. Metrics can be modified or replaced and the IBI remains functional.

Beginning in 2010, the South Dakota Department of Environment and Natural Resources (SD DENR) began implementing biological monitoring on wadeable streams by developing a fish IBI for the Northern Glaciated Plains (NGP) ecoregion in eastern SD (Krause et al. 2013). Prior to this survey the biotic condition of the majority of SD's streams was unknown. The first objective was to develop a fish IBI to serve as the biomonitoring tool kit for western SD prairie stream water quality assessment. The work presented herein will be an expansion on this previous multimetric index from the NGP ecoregion into the NWGP ecoregion of western SD. Because of strong breaks in species ranges at the Missouri River, it is not possible to apply the same metrics identified from the NGP in the NWGP. Additionally, the same metrics might not be as sensitive to regional anthropogenic disturbance, i.e. a species might be more sensitive to sedimentation from a high agriculture setting and less sensitive to fluctuations in stream discharge from storm water runoff in a city. Another difference between the IBI development process in the NWGP and the NGP was that we identified regional candidate reference sites by applying statistical distributions defined from field data and multivariate discriminant analysis and ATtILA to validate those candidate reference sites, rather than using the best professional judgment method.

Methods

Site selection

The NWGP ecoregion comprises nearly one-half of South Dakota's surficial drainage area and is located entirely west of the Missouri River (Fig. 2-1; Bryce et al. 1998). The ecoregion includes ten Level IV ecoregions, eight of which were sampled. The Forested Buttes and the Dense Clay Prairie Level IV ecoregions were eliminated, the former being high gradient rainwater runoff gullies and the latter lacking sufficient perennial wadeable streams to comprise a statistical average. Climate within this ecoregion is semiarid and natural vegetation is primarily mixed and short grass prairie species (Bryce et al. 1998; Chapman et al. 2001). Soils within this ecoregion are derived from shale, siltstone and sandstone (Bryce et al. 1998). Topography is generally flat to rolling, although areas of buttes, badlands and river breaks provide greater relief (Bryce et al. 1998). Much of the ecoregion is managed for cattle grazing, but spring wheat and alfalfa are also common crops (Bryce et al. 1998; Chapman et al. 2001; U.S. Environmental Protection Agency 1996). Larger areas of native grasslands are present. Agriculture is limited by erratic precipitation, which ranges between 33 to 43cm (Chapman et al. 2001).

Sample reaches were selected at random from a target population of over 7,000 wadeable perennial stream segments throughout South Dakota's portion of the NWGP ecoregion. Sixty five sites were identified to represent a random sample of sites spread across the extent of the Level III ecoregion. Those sites were stratified by Level IV ecoregion, which is to say that the number of sites in each ecoregion was proportional to the number of river kilometers in that ecoregion. Sites located immediately below an impoundment or natural basin (5 km buffer) were excluded and all sites were located a minimum of 100m from a road crossing or aquatic barrier. For selected sites where we could not obtain permission from the land owner, another site was chosen at random to replace it. This provided us with a probability-based random sampling of wadeable stream sites, allowing for characterization of stream condition within each LIV ecoregion and across the NWGP as a whole.

Field data collection

Samples were collected following Standard Operating Procedures (SOP) for Field Samplers, Volume II, Biological and Habitat Sampling (SD DENR 2005) once during each growing season from June to August in both 2014 and 2015. All sites were sampled below bankfull conditions to provide a more accurate assessment of normal conditions. Prior to beginning sampling at each site, we established a total reach length and transect spacing. This task was accomplished by measuring the wetted width at ten locations within the target segment. Those ten measurements were averaged to ascertain the preliminary mean stream width (PMSW). If the PMSW was less than or equal to 10 m, transects were spaced three PMSWs apart. If the PMSW was greater than 10 m, transects were spaced two PMSWs apart. The total number of transects at each site were eleven, with transect number eleven always residing upstream and transect number one downstream. We instituted upper and lower limits to reach lengths for very narrow and very wide wadeable streams: minimum of 100 m and maximum of 300 m. Variables linked to water quality criteria in support of beneficial stream uses in South Dakota were measured from each of the sixty five target reaches. Water quality grab samples and multiparameter probe measurements (YSI 556) were collected upstream from transect eleven within each sampled stream reach. During the collection of water-quality samples, instantaneous discharge was also measured at transects one, six, and eleven. A minimum of ten percent of the water quality samples collected were checked for quality assurance and quality control (QA/QC). All water quality samples collected followed the methods outlined in Standard Operating Procedures for Field Samples Volume 1 Tributary and In-Lake Sampling Techniques (SD DENR Water Resources Assistance Program, 2005).

Prior to macroinvertebrate collection, which immediately followed water chemistry sampling, block nets were set at transects 11 and 1 to establish barriers to prevent fish escapement while other biotic and abiotic assessments are taking place in, out, and around target reaches. Fish were collected after other biological samples but before the physical habitat assessment so as to minimize disturbance to the fish community prior to sampling. We collected fish by either seining or electrofishing, depending on the stream channel conditions and water conductivity. If the stream channel contained significant obstructions, such as aquatic vegetation or large rocks, we used the electrofishing method. If the conductivity exceeded limits, electrofishing was ineffective, and we seined. With either method, a single pass was conducted. This was completed in an upstream direction for electrofishing and downstream for seining. Every effort was made to collect fish observed from all habitat types available within the sample reach. In very small streams (<2 m wide) it was possible to sample most of the available habitat, but in larger streams, a meander between habitat types was made in an up or down stream direction depending on fish sampling gear. Two to three personnel conducted the survey, depending on the method used. When using the electrofishing method, one person carried the backpack unit and operated the anode, and another person netted fish. When using the seining method, two people held either end of the net, and a third person lifted the net over any obstructions encountered along the stream reach. Fish survey results were recorded on a data sheet, including the specimen identification to species. Length measurements of the first 100 individuals of each species and counts thereafter, counts were generally made in the field as samples were drawn from field gear. However, some species and small specimens required transport back to the laboratory for closer inspection. Fish less than approximately 25mm total length were not counted as part of the catch. We minimized handling stress by using a portable live well during collection, quickly sorting fish into wet containers, and replacing their water supply. All fish that were alive after processing were immediately returned to the stream, unless they were needed as voucher specimens. For fish that were identified with certainty to species level, two voucher specimens of each fish species were preserved in 10% formalin solution and were retained for quality control and assurance purposes and deposition into the Willis Fish Museum at SDSU. All label voucher containers externally and internally with the site number, sampling date, and species name. Fish that were unidentifiable in the field were euthanized with MS-222 and preserved in formalin for identification in the laboratory.

Detailed physical habitat measurements were taken from each site following collection of water chemistries and biological samples (SD DENR Water Resources Assistance Program, 2005). Habitat data were collected from the entire sample reach and eleven equally spaced transects placed at equidistant locations along the reach. On either end of a transect the riparian land use, dominant vegetation type, animal vegetation use, dominant bank substrate, and bank slumping (presence/absence) was recorded. At eight locations across each transect bed substrate measurements were collected. Stream bank and riparian features were measured with Global Positioning System (GPS) surveying equipment. Several measurements of the channel cross-section were collected to estimate stream width, depth, channel bottom and top width, water depth, channel slope, bank length, bank angle, bank height, bankfull width, bankfull depth, flood prone width, and width:depth ratio. Length of the banks that are vegetated, erosional or depositional, as well as horizontal length of over-hanging vegetation and undercut banks extending over the stream channel bed were also measured. Measures of canopy cover were collected from six stations at each transect using a densiometer. Finally, the number of large woody debris (LWD) was tallied for the entire reach. Length and diameter of all pieces of LWD (> 5 cm diameter) were measured to calculate the volume of LWD within the reach and recorded with the nearest transect.

Statistical Analysis

Counts of individual fish taxa were used to estimate assemblage characteristics (i.e., metrics) which in turn were used to generate community indices of biotic integrity (e.g., Barbour et al. 1999; Whittier et al. 2007). Metrics were compiled through a literature review, and we classified adult fish from our regional species pool into habitat, reproductive, life history, tolerance, and alien guilds (Meador and Carlisle 2007; Whitttier et al. 2007). Fish species were categorized into tolerant ($x \geq 9.2$), moderately tolerant ($9.2 > x > 6.1$), and intolerant ($x \leq 6.1$). Values were calculated by taking tolerance values from Wittier et al. (2007a) and classifying values into the 1st through 15th (intolerant), 16th through 84th (moderately tolerant), and 85st through 99th (tolerant) (Meador et al., 2008; Whittier and Hughes, 1998). Metrics belonged to one of nine metric classes: habitat, tolerance, trophic, reproductive, composition, richness, life history, aliens, and abundance.

We evaluated the full set of candidate metrics with a stepwise screening process (Whittier et al. 2007b). Metrics were eliminated from successive test if they do not pass the previous test. The first step in the screening process was a range test. Metrics were removed if more than 75% of the metric values were the same. The second step in the screening process was a signal to noise test (S:N), which is a statistical approach to classifying the precision and accuracy of sampling and metric analysis. Signal to noise is the ratio of variance between different sites and the variance of repeated sampling of the same site. Metrics were eliminated if there was a S:N score of less than three (Whittier et al. 2007b). The third step was to test for correlation with natural gradients. Abiotic relationships such as stream size, stream slope, and elevation can obscure potential anthropogenic disturbance. Candidate metrics were assessed to account for these natural gradients. All values were regressed against natural gradients, if there were no overlapping values at the ends of the prediction interval a strong relationship was assumed. Metrics were corrected by calculating the offset (analogous to the residual), and these corrected metrics replaced the original metrics (Whittier et al. 2007b). The fourth step was to test for responsiveness to human disturbance, using a one-way analysis of variance. The highest F-statistic from each class of metrics (habitat, tolerance, trophic, reproductive, composition, richness, life history, aliens, and abundance) and metrics with the highest overall significant F-statistic were carried over into the next step of evaluation, provided they were not redundant (Whittier et al. 2007b).The fifth step was to eliminate redundant metrics. Any pair of metrics within a metric class with a Spearman correlation coefficient of greater than 0.70 was considered to be redundant, and the metric with the highest significant F-value from the responsiveness step was retained (Whittier et al. 2007). The final test was a range test for metric scores. Box plots of the metric values were produced for all of the random, least disturbed, and most disturbed sites. If these plots indicated that the majority of the sites had the same metric scores regardless of disturbance class, the metric was eliminated and replaced with a metric that was then next most responsive and was not redundant (Whittier et al. 2007b). Resulting metrics were scored on a continuous scale from 0 to 10 (Bramblett et al. 2005; Hughes et al. 1998; McCormick et al. 2001; Minns et al. 1994). The 5th and 95th percentiles were set as the floor and ceiling values respectively for all sites. Positive metrics in the 95th percentile received a score of 10, the metrics in the 5th percentile received a score of 0, and scores were assigned linearly for all metrics that fell between the 5th and 95th percentiles. All negative metrics were scored similarly except that we calculated the inverse of all values, so that the 5th percentile scored 10 and the 95th percentile scored 0 (Whittier et al. 2007b). Metrics passing this screening process were used to estimate assemblage-specific and integrated indices of biotic integrity. Final IBI scores were scaled 0-100 and expressed as a percent. Sites scoring near 100% are considered high integrity sites.

Reference Site Analysis

In the development of an IBI, the current condition of a body of water must be compared to a reference condition (Stoddard et al. 2006). Historically, and in development of the IBI for the NGP, the reference condition or minimally disturbed site, was set *a priori* (Wang et al. 2003), using best professional judgment. In this study we used a probability based assessment of reference condition (Stoddard et al. 2006; Whittier et al. 2007), using available water quality data and prior evidence of impairment with ATtILA scores. Final IBI scores were scaled 0-100 and expressed as a percent. Sites that scored near 100% were considered high quality sites with respect to biotic integrity. A similar process was followed to screen and score stream sites based upon habitat measurements. Identification and validation of candidate reference sites included statistical analysis of water quality, habitat and IBI data to identify candidate reference sites, validating reference site selections using ATtILA watershed condition scores and multiple discriminant analyses. Candidate reference sites were selected from the upper 10th percentile of sampled sites based on Attila based and watershed condition scores assemblage IBI score distributions. Candidate reference sites were those sites falling within the upper 10th percentile. Scores were generated based on the sum of contributing metric scores and rescaled to fall between 0 (lowest score) and 100 (highest score). We expected candidate reference sites to have watershed condition scores in the upper 75th percentile of their respective Level IV ecoregion. Sites falling below that threshold were rejected as reference sites. We assigned sites to stream condition classes based upon watershed condition scores and indices of biotic integrity. Classes (1-4) were assigned based upon the quartile position of the respective stream site. We used assemblage counts by taxon in a multiple discriminant analysis to evaluate class assignments. We expected discriminant analysis of class assignment to generate high agreement with IBI and habitat data sets. Those candidate reference sites displaying disagreement in site class assignment were rejected and randomly replaced with another candidate reference site falling within the upper 10th percentile (as above) and the process was repeated until all sites falling within the upper 10th percentile of score distributions were evaluated.

Results

In the summers of 2014 and 2015, 65 individual sites were sampled. Fifty-six sites were sampled with repeat visits from June to August in both summers, resulting in 121 total sampling events between the two sampling seasons. After removing one site where no fish were captured. A total of forty one fish species from eleven families were identified from the 39,463 individuals sampled. As a result of naturally depauperate fish communities; sites with low abundances and species counts were retained for analysis.

Six metrics form six different metric classes passed the screening process from the initial 219 candidate metric pool (Table 2-1; Fig. 2-2; Supplementary Appendix). These metrics represented both positive and negative interactions. The positive indicator metrics were Cyprinid Invertivore Species Richness, Proportion of Individuals that are Native Large River Migrants, Proportion of Individuals that are Longnose Dace (LOD) (*Rhinichthys cataractae*), and Proportion of All Species that are Lithophilic Spawners.

The negative indicators of condition were the Proportion of All Species that are Tolerant and the Abundance of Alien Fish.

The range test eliminated eighty two metrics and signal to noise removed the most metrics (103). No metrics were removed or adjusted from the candidate pool during the natural gradient step. Cyprinid Invertivore Species Richness was the most responsive metric ($F_{1,21}$ =20.38, $P < 0.01$). The Proportion of Individuals that are Native Large River Fishes metric was comprised of Channel catfish (*Ictalurus punctatus*), Flathead chub (*Platygobio gracilis*), Freshwater drum (*Aplodinotus grunniens*), Goldeye (*Hiodon alosoides*), Plains Minnow (*Hybognathus placitus*), Spottail shiner (*Notropis hudsonius*), Sturgeon chub (*Macrhybopsis gelida*), Walleye (*Sander vitreus*), and Western Silvery Minnow (*Hybognathus argyritis*). This metric represents the Habitat class, was the second most responsive metric ($F_{1,21}$ =12.51, $P<0.00$), and responded negatively with increasing anthropomorphic disturbance. Abundance of Alien Fish was the only Alien class metric to pass the screening process.

Correlation with natural gradient was assessed by plotting the individual metric values against bankfull width. Bankfull width was used because there were no statistical difference between mean slopes, and watershed size was highly variable ($\bar{x}$ = 1,087.24 km, ± 2,700.31). Only two metrics were correlated with bankfull width, proportion of individuals that are native migrating and proportion of individuals that are native non-tolerant migrating that are intolerant and moderately tolerant species. Those metrics were corrected before continuing with subsequent metric selection. The responsiveness test removed twelve metrics with twenty one metrics in six classes to select candidate metrics. By taking the metric in each class with the highest significant ($P>0.1$) F-value, six metrics were assessed for correlation (Spearman correlation coefficients > 0.7) this step removed fifteen metrics, leaving six metrics across six taken from six metric classes.

Data for two of these metrics (Proportion of All Species that are Tolerant and Proportion of all species that are Native Lithophilic Spawners) are nearly normally distributed across, good, bad, and random sites, whereas scores for the other four metrics tend to be skewed toward smaller values. The least disturbed sites scored twice as high as random sites in the Cyprinid Invertivore Species Richness metric, and the most disturbed sites tended to have no invertivorous cyprinids. Similarly, the Proportion of Individuals that are Longnose Dace (LOD) and the Proportion of Individuals that are Native Large River Migrants one and three sites respectively that had and of those fish at most disturbed sites. Also only one of the least disturbed sites had any Alien Fish (Fig. 2-2)

Each of the six metrics were able to significantly distinguish between least disturbed and most disturbed sites. Cyprinid Invertivore Species Richness ($F_{1,21}$ = 20.38, P < 0.00), Proportion of Individuals that are Native Large River Migrants ($F_{1,21}$ = 12.51, P < 0.001), Proportion of all Species that are Tolerant ($F_{1,21}$ = 3.43, P < 0.05), Abundance of Alien Fish ($F_{1,21}$ = 3.24, P < 0.05), Proportion of All Species that are Lithophilic Spawners ($F_{1,21}$ = 3.36, P < 0.05), and Proportion of Individuals that are Longnose Dace ($F_{1,21}$ = 3.08, P < 0.05).

Seventy-five percent of the least disturbed sites scored between 60 and 85 out of 100 in the IBI, whereas over eighty percent of the most disturbed sites scored between 10 and 45 (Fig. 2-3). Random site IBI scores ranged from under 10 to 90 (Fig. 2-3). There was a significant statistical difference between scores of least disturbed sites and most disturbed sites ($F_{1,21}$ = 27.21, P < 0.00) with least disturbed sites scoring 50% higher on average than most disturbed sites ($\bar{x}$=62.22 ± 5.06; $\bar{x}$=31.36 ± 2.77) (Fig. 2-3). The lowest average IBI scores were in the Missouri Plateau ($\bar{x}$=33.52 ± 7.33), Moreau Prairie ($\bar{x}$=40.83 ± 3.46), River Breaks ($\bar{x}$=35.29 ± 1.97), and Subhumid Pierre Shale Plains ($\bar{x}$=30.83 ± 3.49) ecoregions (Fig. 2-4). Compared to the highest scoring ecoregions the Keya Paha Table Lands ($\bar{x}$=56.31 ± 4.57), Sage Brush Steppe ($\bar{x}$=60.33 ± 4.99), and the White River Bad Lands ($\bar{x}$=67.22 ± 4.16).

Discussion

Sequential filtering of metrics resulted in six metrics representing six different metric classes. All metrics showed a significant difference between least and most disturbed sites: Fish metrics that required a long life history failed the screening process, as a result of the low numbers of long lived species captured. Also, within the Life History class any metrics pertaining to fish requiring long migrations failed to pass the screening process. No metrics within the Richness and Abundance metric classes passed the signal: noise test indicating that there was high variability between sampling events with those specific metrics. The lack of responsiveness of these metrics was expected as Krause et al. (2013) found similar results when analyzing data from the NGP III ecoregion. Much of the variation can be attributed to inconsistency of catch rates within target reaches (Hughes and Oberdorff, 1999).

The Sagebrush Steppe, Semiarid Pierre Shale Planes, White River Badlands, and Keya Paha Tablelands ecoregions held the highest average site scores. This could be a result of the lower density of row crop agriculture within these ecoregions; where differences in soil, topography, and climate make these ecoregions more suitable to cattle and sheep grazing. We observed little effect of population density on IBI scores, because outside of a few metropolitan areas in the entirety of Western South Dakota, population density is very low. IBI scores also responded similarly to increases in row crop agriculture; where we observed higher densities in crop land IBI average scores were poorest.

Cyprinid Invertivore Species Richness from the Trophic class was an adaptation from Karr's (1981) original metric, Proportion of insectivorous Cyprinids as a means to assess invertebrate communities. Evaluating cyprinid invertivores is valuable, as Hughes and Oberdorff (1999) found, within the U.S. and Canada invertivore species dominate most streams. Karr (1981) found loss of prey base (invertebrates in this case) is a measure of both degradation of water quality and or habitat loss. He continues by stating that there exists a relationship where a high number of cyprinid invertivores and a low abundance of omnivores generally resulted in a better stream condition (Karr 1981). As a result of this research and when compared with box plots we found an increase in cyprinid invertivore species richness to be more correlated with sites with less disturbance.

When compared with reference sites the Proportion of Individuals that are Native Large River Fishes metric showed a large range from 52 % to zero, but no "most disturbed" site had more than 10 % native large river fish. These fishes represent an assemblage that has adapted to the harsh climate of Western South Dakota and require streams and rivers free of obstructions that would impede fish movement i.e. large dams and reservoirs and drop culverts (U.S. Fish and Wildlife Service. 1993; U.S. Fish and Wildlife Service. 2001; Rahel and Thel, 2004a; Rahel and Thel, 2004b). This metric forces us to change the paradigm of how we view "good" streams within the North Great

Plains ecoregion. Most of the fish that form this assemblage of native large river fishes are highly efficient at surviving in turbid waters, changes in water clarity may be more aesthetically pleasing to the general public but causes a releases of the competitive advantage these native fish species have (U.S. Fish and Wildlife Service. 1993; U.S. Fish and Wildlife Service. 2001; Rahel and Thel, 2004a; Rahel and Thel, 2004b).

The Proportion of All Species that are Tolerant metric was the most responsive metric from the Tolerance class. This metric represents one of the two negatively correlated metrics that passed the screening process, the other being Abundance of Alien Fish. It is classified as negative because it is calculated based off the most tolerant species within the NWGP assemblage which include Black Bullhead (*Ameiurus melas*), Brassy Minnow (*Hybognathus hakinson*), Brook Stickleback (*Culaea inconstans*), Freshwater Drum, Goldeye, and Orangespotted Sunfish (*Lepomis humilis*). This is another metric that is well represented in the literature, whether abundance, proportion, or species richness some assessment of the tolerant fish has shown to be an appropriate measure of anthropogenic disturbance. Multiple studies have shown that an increase in tolerant fish is correlated with an increase in anthropogenic disturbance, or a decrease in IBI score (OEPA 1987; Crumby et al., 1990; Bramblett and Fausch, 1991; Simon, 1991, 1992; Hoefs and Boyle, 1992; Lyons, 1992; Goldstein et al., 1994; ; Bramblett et al., 2005).

In the Reproductive class of metrics the metric of the Proportion of All Species that are Native Lithophilic Spawners passed the screening processes. The proportion of lithophilic spawning fish is an indicator of non-point source pollution as these fish require the interstitial spaces of sand and larger substrates free from silt. This metric can be used to assess habitat degradation of lands surrounding streams and has become an influential metric, specifically in the Midwest where row crop agriculture dominates the landscape. The benefits of using a lithophilic metric is evident in the selectability of this metric in regional IBIs across the U.S. (OEPA 1987; Hoefs and Boyle, 1992; Lyons, 1992; Simon, 1992; Bailey et al., 1993; Goldstein et al., 1994; Whittier et al. 2007b; Krause et al., 2013;).

There was also low abundance of alien fish represented in this study, but due in part to the ability of Common Carp (Cyprinus carpio) to effectively colonize degraded systems and the compounding effect nonnative carp have in further degrading the system (increasing turbidity, uprooting vegetation, sediment resuspension (Pascal 2015) the metric was retained for South Dakota's Northwestern Great Plains ecoregion IBI. This metric was negatively associated with disturbance class and was represented in the scoring as a negative indicator or stream health.

Proportion of Individuals that are *Rhinichthys* species, which was changed to Longnose Dace (no other *Rhinichthys* species were captured) which was taken from Steedman (1988) and represents the Composition metric class. Steedman (1988) found that *Rhinichthys* species (Blacknose Dace (*Rhinichthys atratulus*) and Longnose Dace) increased with increased anthropogenic disturbance in southern Ontario. The range test for the NWGP IBI showed that Longnose Dace were more associated with the least disturbed sites. This was contrary to Steedman's results, but Longnose Dace ecology parallels that of other metrics selected for this ecoregion. Longnose Dace spawn on coarser gravels and overhead cover (Edwards et al. 1983). They can also tolerate turbid waters (Edwards et al. 1983) and were found to be moderately tolerant within the assemblage of the NWGP of South Dakota. As such, this metric was classified as a positive metric and was scored accordingly.

When comparing the selected metrics form SD's NWGP to Whittier (2007b) in the plains ecoregion which included South Dakota, North Dakota, and parts of Montana, Wyoming, and Colorado the metric classes were similar. These regional IBIs resulted in Lithophilic spawers for the Reproduction class, alien species, and invertivores in the Trophic class. The NWGP project did not have a richness metric pass the screening process, as stated above, and our composition metric was different; where Whittier (2007b) selected proportion of Ictaluridae, the NWGP's most responsive composition metric was Proportion of individuals that are *Rhinichthys* species. The metric proportion of vertebrate abundance in the family *Ictaluridae* failed to pass the S:N test early in metric selection, but channel catfish our most abundant species in the *Ictaluridae* family was represented in the proportion of individuals that are native large river metric.

Conclusion

The methods Whittier et al. (2007b) have provided have shown that in the NWGP we are able to make the distinction between sites with increased anthropogenic disturbance and those with less disturbance. The NWGP fish IBI will serve as a baseline for continuing monitoring of anthropogenic disturbance in western SD. In addition, the conclusions have led to classification of reference sites to serve future monitoring in the region. This will provide a benchmark for monitoring as SD tracks climate change, increases in agricultural activity, and growing populations in western SD.

Literature Cited

Balvanera, P., A. B. Pfisterer, N. Buchmann, J. He, T. Nakashizuka, D. Raffaelli, and B. Schmid 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters 9:1146-1156

Barbour, M. T., J. Gerritsen, B. D. Synder, and J. B. Stribling, 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish. Office of Water, editor. US Environmental Protection Agency, Washington D.C.

Barbour, M. T., J. B. Stribling, and J. R. Karr. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. Lewis Publishers, Boca Raton, Florida.

Berka C., H. Schreier and K. Hall 2001. Linking water quality with Agricultural Intensification in a Rural Watershed. Water, Air, and Soil Pollution 127: 389–401.

Bramblett, R.G., Fausch, K.D., 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river. Trans. Am. Fish. Soc. 120: 752-769.

Bramblett, R. G., T. R. Johnson, A. V. Zale, and D. G. Heggem. 2005. Development and evaluation of a fish assemblage index of biotic integrity for northwestern Great Plains streams. Transactions of the American Fisheries Society 134(3):624-640.

Bryce, S., and coauthors. 1998. Ecoregions of North Dakota and South Dakota. Northern Prairie Wildlife Research Center Online, Jamestown, ND.

Chapman, S. S., J. M. Omernik, J. A. Freeouf, D. G. Huggins, J. R. McCauley, C. C. Freeman, G. Steinauer, R. T. Angelo, and R. L. Schlepp. 2001. Ecoregions of Nebraska and Kansas (color poster with map, descriptive text, summary tables, and photographs): Reston, Virginia, U.S. Geological Survey (map scale 1:1,950,000).

Crumby, W.D., Webb, M.A., Bulow, F.J., Cathey, H.J., 1990. Changes in the biotic integrity of a river in north-central Tennessee. Trans. Am. Fish. Soc. 119: 885-893.

Diana, M., J. D. Allan, D. Infante. 2006. The Influence of Physical Habitat and Land Use on Stream Fish Assemblages in Southeastern Michigan. American Fisheries Society Symposium 48: 359-374

Dudgeon, D. 1992. Endangered ecosystems - a review of the conservation status of tropical Asian rivers. Hydrobiologia 248:167-191.

Edwards, E. A., H. Li, C. B. Schreck. 1983. Habitat suitability index models: Longnose dace. U.S. Dept. Int., Fish and Wildlife Service. FWS/OBS-82/10.33. 13 pp.

Fausch, K. D., J. R. Karr, and P.R. Yant. 1984. Regional applications of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society 113:39-55.

Frey, D. G. 1977. Biological integrity of water: an historical approach. Pages 127-140 in R. K. Ballantine, and L. J. Guarraia, editors. The integrity of water. U.S. Environmental Protection Agency, Washington D.C.

Gleick, P.H. 1996. “Basic water requirements for human activities: Meeting basic needs.” Water International 21(2):83-92.

Gleick, P. H., and M. Palaniappan. 2010. Peak water limits to freshwater withdrawal and use. Proceedings of the National Academy of Sciences of the United States of America 107(25):11155-11162

Goldstein, R.M., Simon, T.P., Bailey, P.A., Ell, M., Schmidt, K., Emblom, J.W., 1994. Proposed metrics for the index of biotic integrity for the streams of the Red River of the North basin, Proceedings of the North Dakota Water Quality Symposium, Fargo, ND, pp. 169-180.

Gorman, O.T., and J. R. Karr 1978. Habitat Structure and Stream Fish Communities. Ecology 59:507–515

Hoefs, N.J., Boyle, T.P., 1992. Contribution of fish community metrics to the indices of biotic integrity in two Ozark rivers, in: McKenzie, D.H., Hyatt, D.E., MacDonald, V.J. (Eds.), Ecol. Indic. Elsevier Applied Science, New York, pp. 283-303.

Hoeinghaus DJ, Winemiller KO, Birnbaum JS. 2007. Local and regional determinants of stream fish assemblage structure: inferences based on taxonomic vs. functional groups. Journal of Biogeography 34: 324-338.

Hughes, R. M., and coauthors. 1998. A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Sciences 55(7):1618-1631.

Hughes, R.M., Oberdorff, T., 1999. Applications of IBI concepts and metrics to waters outside the United States and Canada, in: Simon, T.P. (Ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL, pp. 79-96.

Jackson, D. A., P. R. Peres-Neto, and J. D. Olden. 2001. What controls who is where in freshwater fish communities-the roles of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58:157–170.

Karr, J. R. 1981. Assessment of Biotic Integrity Using Fish Communities. Fisheries 6(6) 21-27

Karr, J. R. 1987. Biological monitoring and environmental assessment: a conceptual framework. Environmental Management 11(2):249-256-256.

Karr, J. R. 1993. Defining and assessing ecological integrity: Beyond water quality. Environmental Toxicology and Chemistry 12(9):1521-1531.

Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington D. C.

Karr, J. R., and D. R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management 5(1):55-68.

Krause, J. R., K. N. Bertrand, A. Kafle, and N.H. Troelstrup. 2013. A fish index of biotic integrity for South Dakota's Northern Glaciated Plains Ecoregion. Ecological Indicators, 34:313-322.

Lobb, M. D. III, and D. J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. Transactions of the American Fisheries Society 120:65–78.

Loomis, J., P. Kent, L. Strange, K. Fausch, and A. Covich. 2000. Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. Ecological Economics 33(1):103-117.

Lyons, J., 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149. U. S. Department of Agriculture, St. Paul, MN.

Marsh-Matthews, E., and W. J. Matthews. 2000. Geographic, terrestrial and aquatic factors: which most influence the structure of stream fish assemblages in the midwestern United States? Ecology of Freshwater Fish 9:9–21.

Meador, M. R., and D. M. Carlisle. 2007. Quantifying tolerance indicator values for common stream fish species of the United States. Ecological Indicators 7(2):329-338.

Olden, J. D., and D.A. Jackson. 2001. Fish–habitat relationships in lakes: gaining predictive and explanatory insight by using artificial neural networks. Transactions of the American Fisheries Society, 130(5):878-897.

Badiou, Pascal H. J., L. G. Goldsborough. 2015. Ecological impacts of an exotic benthivorous fish, the common carp (Cyprinus carpio L.), on water quality, sedimentation, and submerged macrophyte biomass in wetland mesocosms. Hydrobiologia, 755(1):107-121

Poff, N. L., and coauthors. 1997. The natural flow regime. Bioscience 47(11):769-784.

Rahel, F.J. and L.A. Thel. 2004a. Flathead Chub (Platygobio gracilis): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/

Rahel, F.J. and L.A. Thel. (2004b). Sturgeon Chub (Macrhybopsis gelida): a technical conservation assessment. [Online]. USDA Forest Service, Rocky Mountain Region. Available: http://www.fs.fed.us/r2/projects/scp/assessments/sturgeonchub.pdf [date of access]. assessments/flatheadchub.pdf [date of access].

Rich, C. F., T. E. McMahon, B. E. Rieman, and W. L. Thompson. 2003. Small habitat, watershed, and biotic features associated with bull trout occurrence in Montana streams. Transactions of the American Fisheries Society 132:1053–1064.

Ruaro, R., and E.A. Gubiani. 2013. A scientometric assessment of 30 years of the Index of Biotic Integrity in aquatic ecosystems: Applications and main flaws. Ecological Indicators, 29:105-110.

Schlosser, I. J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52:395–414.

Simon, T.P., 1991. Development of ecoregion expectations for the index of biotic integrity: central corn belt plain. U.S. Environmental Protection Agency, Chicago.

Simon, T.P., 1992. Development of biological criteria for large rivers with an emphasis on an assessment of the White River drainage, Indiana. U.S. Environmental Protection Agency, Chicago.

Sindt, A.R., M.C. Quist, and C.L. Pierce. 2012. Habitat associations of Fish species of greatest conservation need at multiple spatial scales in wadeable Iowa streams. North American Journal of Fisheries Management 32: 1046-1061.

South Dakota DENR. 2005. Standard operating procedures for field samplers. Department of Environment and Natural Resources, editor, Pierre, SD.

Steedman, R.J., 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Can. J. Fish. Aquat. Sci. 45:492-501.

Stoddard, J. L., D. P. Larsen, C. P. Hawkins, R. K. Johnson, and R. H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications 16(4):1267-1276.

Stoddard, J. L., and coauthors. 2005. Environmental monitoring and assessment program (EMAP): western streams and rivers statistical summary. U.S. Environmental Protection Agency, editor, Washington D.C.

U.S. Environmental Protection Agency, 1996. Level III ecoregions of the continental United States (revision of Omernik, 1987): Corvallis, Oregon, U.S. Environmental Protection Agency - National Health and Environmental Effects Research Laboratory Map M-1, various scales

U.S. Environmental Protection Agency. 2004. Wadeable stream assessment: benthic laboratory methods. U.S. Environmental Protection Agency, editor. Office of Water and Office of Research and Development,, Washington D.C.

U.S. Fish and Wildlife Service. 1993. Status report on sturgeon chub (Macrhybopsis gelida), a candidate endangered species. U.S. Fish and Wildlife Service, Bismarck, North Dakota. 58 pp.

U.S. Fish and Wildlife Service. 2001. Updated Status Review of Sicklefin and Sturgeon Chub in the United States. U.S. FISH AND WILDLIFE SERVICE, Denver, Colorado

Wang, L., J. Lyons, P. Rasmussen, P. Seelbach, T. Simon, M. Wiley, P. Kanehl, E. Baker, S. Niemela, and P.M. Stewart. 2003.Watershed, reach, and riparian influence on stream fish assemblages in the northern lakes and forest ecoregion, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 60:491–505.

Waters, C. N., J. Zalasiewicz, C. Summerhayes, A. D. Barnosky, C. Poirier, A. Gałuszka, A. Cearreta, M. Edgeworth, E. C. Ellis, M. Ellis, C. Jeandel, R. Leinfelder, J. R. McNeill, D. d. Richter, W. Steffen, J. Syvitski, D. Vidas, M. Wagreich, M. Williams, A. Zhisheng, J. Grinevald, E. Odada, N. Oreskes & A. P. Wolfe, 2016.

The Anthropocene is functionally and stratigraphically distinct from the Holocene. Science 351(6269) doi:10.1126/science.aad2622.

Whittier, T. R., R. M. Hughes, G. A. Lomnicky, and D. V. Peck. 2007a. Fish and amphibian tolerance values and an assemblage tolerance index for streams and rivers in the western USA. Transactions of the American Fisheries Society 136(1):254-271.

Whittier, T. R., and coauthors. 2007b. A structured approach for developing indices of biotic integrity: Three examples from streams and rivers in the western USA. Transactions of the American Fisheries Society 136(3):718-735.

Table 2-1. Retained Metrics for the South Dakota's Northwestern Great Plains ecoregion IBI. Table is presented in order of responsiveness (F-values) from a one-way ANOVA comparing least disturbed sites to most disturbed sites. S:N ratios compares variance among sites to variance within sites. Floor and ceiling values represent the highest and lowest metric values needed to score 10 or 0 depending on metric response.

Metric	Metric Class	F-value	S:N	Floor	Ceiling	Response
Cyprinid Invertivore Species Richness	TROPHIC	20.4	6.6	0.000	2.000	+
Native Large River [a]	HABITAT	12.5	17.9	0.000	0.503	+
Tolerant [b]	TOLERANCE	3.4	5.1	0.329	0.000	-
Native Lithophilic Spawners [b]	REPRODUCTIVE	3.4	6.0	0.000	0.564	+
Abundance of Alien Fish	ALIEN	3.2	6.2	4.900	0.000	-
Rhinichtys cataractae [a]	COMPOSITION	3.1	6.1	0.000	0.308	+

[a] Proportion of Individuals

[b] Proportion of Taxa

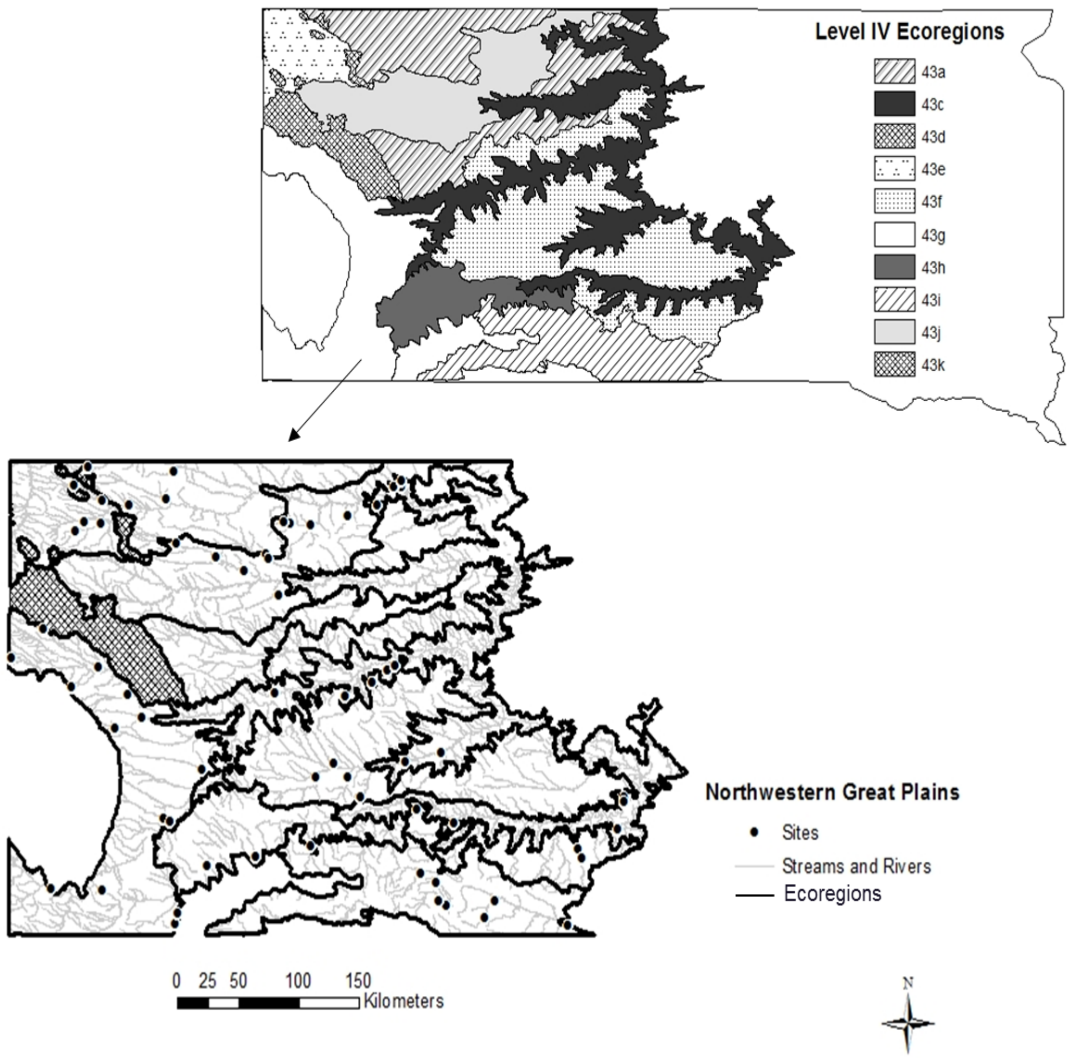

Fig. 2-1. Level IV Ecoregions within the Level III Northwestern Great Plains Ecoregions within the state of South Dakota. Location of 65 reach locations. Ecoregions 43d and 43k are cross hatched and were not sampled in this study due to lack of perennial wadeable streams.

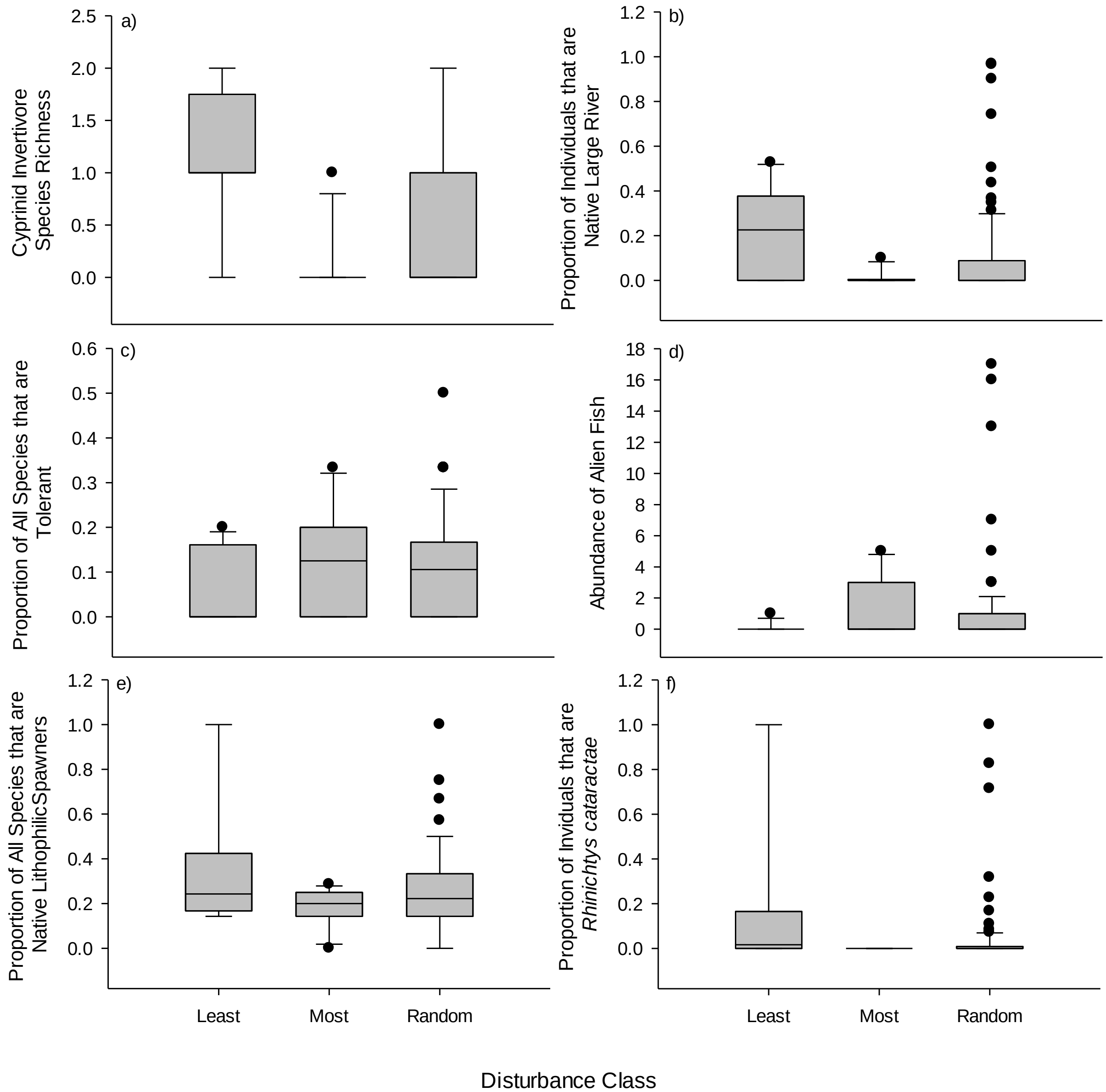

Fig. 2-2. Distribution of retained metric values from the range test from least, most, and random disturbance classes for South Dakota's Northwestern Great Plains ecoregion IBI. Plots show medians and quartiles with whiskers representing 10th and 90th percentiles, black dots show outliers.

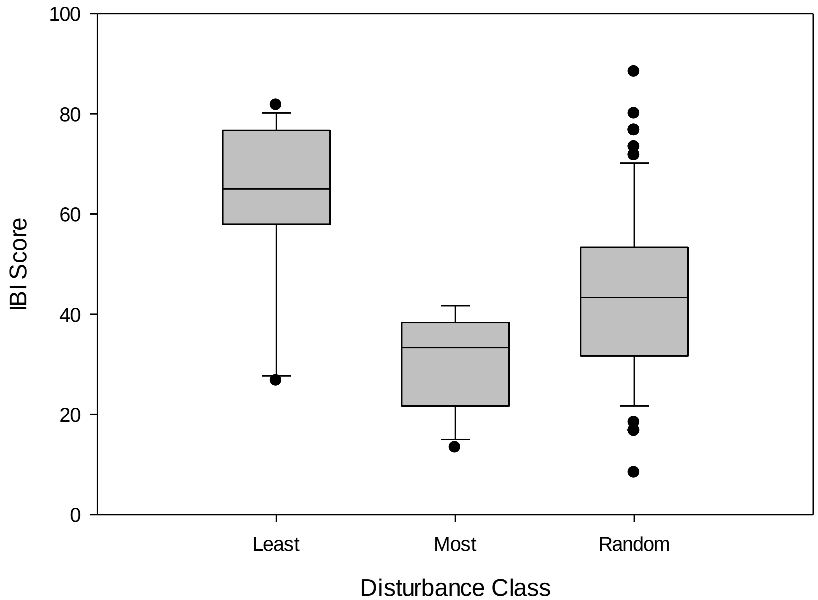

Fig. 2-3. Distribution of IBI scores from each disturbance class within the Northwestern Great Plains ecoregion in South Dakota. Plots show medians and quartiles with whiskers representing 10th and 90th percentiles, black dots show outliers.

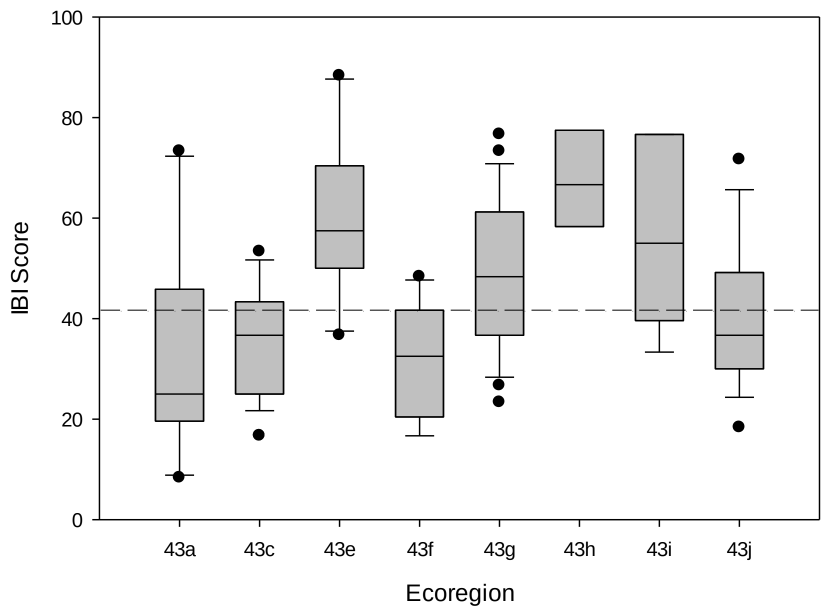

Fig. 2-4. IBI scores by Level IV ecoregions in the Northwestern Great Plains of South Dakota. Plots show medians and quartiles with whiskers representing 10th and 90th percentiles, black dots show outliers.

Supplementary Appendix

Table A.1.—Fish species characteristics used to calculate metrics in South Dakota's Northwestern Great Plains ecoregion. Species are listed alphabetically by common name. Categories as follows: Hab. (preferred habitat; B = benthic, H = hider, WC = water column); Lot. (lotic; X = prefers flowing waters, L = prefers large rivers, R = rheophilic); Drom. (migratory; P = potadromous); Temp. (temperature; W = warm, CL = cool, CD = cold); Troph. (trophic; D = detritivore, H = herbivore, I = invertivore, P = piscivore; O = omnivore which is based on a compilation of the previous trophic feeding guilds); Repr. (preferred reproductive habitat; A11 = pelagophil, A12 = lithopelogophil, A13 = lithophil, A14 = phytolithophil, A15 = phytophil, A23 = lithophil brood hider, A24 = crevice spawner, B = nest guarder, B27 = speleophil; S = serial spawner; E = reproductively mature <2 years, L=reproductively mature >3 years); Long-lived (>8 years); Tol. (tolerance; T = tolerant, M= moderate, I = intolerant), Air (can breathe air); T & E (listed as state threatened, endangered, or of concern) is noted *; Alien (not native to South Dakota wadeable streams) is noted **.

Species	Hab	Lot	Drom	Temp	Trop	Repr	Long Lived	Tol	Air
Bigmouth Shiner *Notropis dorsalis*	B	X		W	O	B		mod	
Black Bullhead *Ameiurus melas*	B,H			W	IP	B27		tol	X
Black Crappie *Pomoxis nigromaculatus*	H,WC			W	IP	B		mod	
Bluegill *Lepomis macrochirus*	H,WC			W	IP	B		mod	
Brassy Minnow	B	X		C	O	A15		tol	

Hybognathus hankinsoni									
Brook Stickleback	H,W C			C	I	B24		tol	
Culaea inconstans									
Brown Trout**	H,W C	R	P	CD	IP	A23	X	int	
Salmo trutta									
Channel Catfish	B,H	L	P	W	IP	B27		mod	X
Ictalurus punctatus									
Common Carp**	B			W	O	A14	X	mod	X
Cyprinus carpio									
Creek Chub	WC	X		C	O	A23		mod	
Semotilus atromaculatus									
Fathead Minnow	WC			W	O	B		mod	
Pimephales promelas									
Flathead Chub	B	L		C	I	A11		mod	
Platygobio gracilis									
Freshwater Drum	B	L		W	IP	A11	X	tol	
Aplodinotus grunniens									
Gizzard Shad	WC			W	O	A14		mod	
Dorosoma cepedianum									
Golden Shiner	WC			W	O	A15		mod	
Notemigonus crysoleucas									
Goldeye	WC	L		C	IP	A12		tol	
Hiodon alosoides									
Green Sunfish	H,W C			W	IP	B	X	mod	
Lepomis cyanellus									
Iowa Darter	B,H	X		C	I	B		mod	
Etheostoma exile									
Johnny Darter	B,H	X		W	I	B		mod	
Etheostoma nigrum									
Largemouth Bass	H,W C			W	P	B	X	int	
Micropterus salmoides									
Longnose Dace	B,H	R		C	I	A12		mod	
Rhinichthys cataractae									
Mountain Sucker*	B	X		C	H	A13	X	int	
Catostomus platyrhynchus									
Northern Pike	H,W C			C	P	A15	X	mod	
Esox lucius									
Orange Spotted Sunfish	H,W C			W	IP	B		tol	
Lepomis humilis									

Plains Killifish *Fundulus zebrinus*	WC			W	I	A15		int
Plains Minnow *Hybognathus placitus*	B	L		W	O	A11		mod
Plains Topminnow* *Fundulus sciadicus*	WC			W	I	A15		int
Red Shiner *Cyprinella lutrensis*	WC	X		W	O	B		mod
River Carpsucker *Carpiodes carpio*		X	P	W	O	A12	X	mod
Rock Bass *Ambloplites rupestris*	H,WC			W	IP	B		mod
Sand Shiner *Notropis stramineus*	WC	X		W	O	A14		mod
Shorthead Redhorse *Moxostoma macrolepidotum*	B	X	P	W	I	A13	X	mod
Smallmouth Bass *Micropterus dolomieu*	H,WC			C	P	B	X	int
Spottail Shiner *Notropis hudsonius*	WC	L		C	O	A12		mod
Stonecat *Noturus flavus*	B,H	X		W	IP	B27		mod
Sturgeon Chub* *Macrhybopsis gelida*	WC	L,R		W	I	A11		mod
Walleye *Sander vitreus*	WC	L		C	P	A12	X	mod
Western Silvery Minnow *Hybognathus argyritis*	B	L		W	O	A11		mod
White Crappie *Pomoxis annularis*	H,WC			W	IP	B		mod
White Sucker *Catostomus commersonii*	B	X	P	C	O	A12	X	mod
Yellow Perch *Perca flavescens*	WC			C	P	A12	X	mod

Table A. 2. Metrics in alphabetic order by their metric class. All metric without references were taken from Whittier (2007b)

Metric Class	Metric
ABUNDANCE	Abundance of all Native Vertebrates
ABUNDANCE	Abundance of Fish
ALIEN	Abundance of Alien Fish
ALIEN	Alien Lotic Species Richness all X
ALIEN	Alien Vertebrate Species Richness
ALIEN	Proportion of All Species that are Alien Lotic all X
ALIEN	Proportion of All Species that are Native Aquatic
ALIEN	Proportion of Fish Species that are Alien
ALIEN	Proportion of Individual Fish that are Alien
ALIEN	Proportion of Individuals that are Alien Lotic all X
ALIEN	Proportion of Individuals that are Native Aquatic
COMPOSITION	Abundance of Native Catostomids and Native Ictalurids
COMPOSITION	Catostomidae Richness (Bailey 1993)
COMPOSITION	Catostomidae Richness minus catcom (Karr 1981)
COMPOSITION	Centrarchidae plus micsal Richness (Simon 1992)
COMPOSITION	Centrarchidae plus perfla and micsal Richness
COMPOSITION	Centrarchidae plus perfla Richness (Lyons 1992)
COMPOSITION	Centrarchidae Richness (Simon 1991)
COMPOSITION	Cyprinidae Richness (Ohio 1989)
COMPOSITION	Cyprinidae Richness minus Cypcar, Sematr, Pimpro (Bailey 1993)
COMPOSITION	Darter Richness (Simon 1991)
COMPOSITION	Dominance (ME) top 2 abundance of species
COMPOSITION	Dominance (Niemela 1999)
COMPOSITION	Dominance (Wilton 2004) top 3 abundance of species
COMPOSITION	Dominance (Wilton 2004) top 5 abundance of species
COMPOSITION	Native Catostomid and Cyprinid Species Richness (Hoefs 1992)
COMPOSITION	Native Catostomid and Ictalurid Species Richness
COMPOSITION	Proportion of flathead chubs
COMPOSITION	Proportion of Inviduals that are Cyprinus carpio (Hughes 1987)
COMPOSITION	Proportion of Inviduals that are Lepomis cyanellus (Karr 1981)
COMPOSITION	Proportion of Inviduals that are Rhinichtys obtusus (Steedman 1988) Change to longnose

COMPOSITION	Proportion of Inviduals that are Semotilus atromaculatus (Leonard 1986)
COMPOSITION	Proportion of Pioneer species (Ohio 1987) ethnig, lepcya, pimpro, pimnot, sematr
COMPOSITION	Proportion of Vertebrate Abundance in the Family Catostomidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Catostomidae minus catcom (Ohio 1987)
COMPOSITION	Proportion of Vertebrate Abundance in the Family Centrarchidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Clupeidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Cyprinidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Esocidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Fundulidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Gasterosteidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Ictaluridae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Percidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Salmonidae
COMPOSITION	Proportion of Vertebrate Abundance in the Family Umbridae
COMPOSITION	Shannon Weaver Diversity Index
HABITAT	Native Benthic Species Richness
HABITAT	Native Coolwater Species Richness
HABITAT	Native Hider Species Richness
HABITAT	Native Large River Species Richness
HABITAT	Native Lotic Species Richness X
HABITAT	Native Rheophilic Species Richness
HABITAT	Native Water Column Species Richness
HABITAT	Proportion of All Species that are Native Benthic
HABITAT	Proportion of All Species that are Native Coolwater
HABITAT	Proportion of All Species that are Native Hider
HABITAT	Proportion of All Species that are Native Large River
HABITAT	Proportion of All Species that are Native Lotic X
HABITAT	Proportion of All Species that are Native Nontolerant Benthic mod + int
HABITAT	Proportion of All Species that are Native Nontolerant Coolwater int + mod
HABITAT	Proportion of All Species that are Native Nontolerant Hider int + mod
HABITAT	Proportion of All Species that are Native Nontolerant Large River int + mod
HABITAT	Proportion of All Species that are Native Nontolerant Lotic X int + mod
HABITAT	Proportion of All Species that are Native Nontolerant Rheophilic int + mod
HABITAT	Proportion of All Species that are Native Nontolerant Water Column int + mod
HABITAT	Proportion of All Species that are Native Rheophilic
HABITAT	Proportion of All Species that are Native Sensitive Benthic int
HABITAT	Proportion of All Species that are Native Sensitive Coolwater int
HABITAT	Proportion of All Species that are Native Sensitive Hider int
HABITAT	Proportion of All Species that are Native Sensitive Large River int
HABITAT	Proportion of All Species that are Native Sensitive Lotic X int
HABITAT	Proportion of All Species that are Native Sensitive Rheophilic int
HABITAT	Proportion of All Species that are Native Sensitive Water Column int
HABITAT	Proportion of All Species that are Native Water Column

HABITAT	Proportion of All Species that are Sensitive Rheophilic int
HABITAT	Proportion of Individuals that are Native Benthic
HABITAT	Proportion of Individuals that are Native Coolwater
HABITAT	Proportion of Individuals that are Native Hider
HABITAT	Proportion of Individuals that are Native Large River
HABITAT	Proportion of Individuals that are Native Lotic X
HABITAT	Proportion of Individuals that are Native Nontolerant Benthic mod + int
HABITAT	Proportion of Individuals that are Native Nontolerant Coolwater int + mod
HABITAT	Proportion of Individuals that are Native Nontolerant Hider int + mod
HABITAT	Proportion of Individuals that are Native Nontolerant Large River int + mod
HABITAT	Proportion of Individuals that are Native Nontolerant Lotic X int + mod
HABITAT	Proportion of Individuals that are Native Nontolerant Rheophilic int + mod
HABITAT	Proportion of Individuals that are Native Nontolerant Water Column int + mod
HABITAT	Proportion of Individuals that are Native Rheophilic
HABITAT	Proportion of Individuals that are Native Sensitive Benthic int
HABITAT	Proportion of Individuals that are Native Sensitive Coolwater int
HABITAT	Proportion of Individuals that are Native Sensitive Hider int
HABITAT	Proportion of Individuals that are Native Sensitive Large River int
HABITAT	Proportion of Individuals that are Native Sensitive Lotic X int
HABITAT	Proportion of Individuals that are Native Sensitive Rheophilic int
HABITAT	Proportion of Individuals that are Native Sensitive Water Column int
HABITAT	Proportion of Individuals that are Native Water Column
HABITAT	Proportion of Individuals that are Sensitive Rheophilic int
HABITAT	Threatened & Endangered Species Richness
HABITAT	Water Column Cyprinid Species Richness (Hoefs 1992)
LIFE HISTORY	Native Long-lived Species Richness
LIFE HISTORY	Native Migrating Species Richness P
LIFE HISTORY	Proportion of All Species that are Native Long-lived
LIFE HISTORY	Proportion of All Species that are Native Migrating P
LIFE HISTORY	Proportion of All Species that are Native Nontolerant Long-lived int + mod
LIFE HISTORY	Proportion of All Species that are Native Nontolerant Migrating P int + mod
LIFE HISTORY	Proportion of All Species that are Native Sensitive Long-lived int
LIFE HISTORY	Proportion of All Species that are Native Sensitive Mirgrating P int
LIFE HISTORY	Proportion of Individuals that are Native Long-lived
LIFE HISTORY	Proportion of Individuals that are Native Migrating P
LIFE HISTORY	Proportion of Individuals that are Native Nontolerant Long-lived int + mod
LIFE HISTORY	Proportion of Individuals that are Native Nontolerant Migrating P int + mod
LIFE HISTORY	Proportion of Individuals that are Native Sensitive Long-lived int
LIFE HISTORY	Proportion of Individuals that are Native Sensitive Migrating P int
REPRODUCTIVE	Abundance of Generalist Spawner Individuals A11
REPRODUCTIVE	Abundance of Lithophilic Individuals A13,A23,A12
REPRODUCTIVE	Abundance of Native Lithophilic Individuals A13,A23,A12
REPRODUCTIVE	Abundance of Non-Lithophilic Nest Guarding Individuals B, B27

REPRODUCTIVE	Abundance of Sensitive Spawner Individuals A23, A24, B27
REPRODUCTIVE	Generalist Spawner Species Richness A11
REPRODUCTIVE	Lithophilic Species Richness A13,A23,A12
REPRODUCTIVE	Native Lithophilic Species Richness A13,A23,A12
REPRODUCTIVE	Proportion of All Species that are Generalist Spawner A11
REPRODUCTIVE	Proportion of All Species that are Lithophil A13,A23,A12
REPRODUCTIVE	Proportion of All Species that are Native Lithophil A13,A23,A12
REPRODUCTIVE	Proportion of All Species that are Non-Lithophilic Nest Guarders B, B27
REPRODUCTIVE	Proportion of All Species that are Sensitive Spawner A23, A24, B27
REPRODUCTIVE	Proportion of Individuals that are Generalist Spawner A11
REPRODUCTIVE	Proportion of Individuals that are Lithophil A13,A23,A12
REPRODUCTIVE	Proportion of Individuals that are Native Lithophil A13,A23,A12
REPRODUCTIVE	Proportion of Individuals that are Non-Lithophilic Nest Guarders B, B27
REPRODUCTIVE	Proportion of Individuals that are Sensitive Spawner A23, A24, B27
RICHNESS	Fish Species Richness
RICHNESS	Native Fish Species Richness
RICHNESS	Native Vertebrate Family Richness
RICHNESS	Native Vertebrate Species Richness
RICHNESS	Native Vertebrate Species Richness
RICHNESS	Non-Lithophilic Nest Guarding Species Richness B, B27
RICHNESS	Non-Tolerant Species Richness int + mod
RICHNESS	Vertebrate Family Richness
TOLERANCE	Air Breathing Species Richness
TOLERANCE	Native Sensitive Species Richness int
TOLERANCE	Proportion of All Species that are Airbreather
TOLERANCE	Proportion of All Species that are Native Sensitive int
TOLERANCE	Proportion of All Species that are Tolerant tol
TOLERANCE	Proportion of Individuals that are Airbreather
TOLERANCE	Proportion of Individuals that are Native Sensitive int
TOLERANCE	Proportion of Individuals that are Threatened & Endangered
TOLERANCE	Proportion of Individuals that are Tolerant tol
TOLERANCE	Sensitive Spawner Species Richness A23, A24, B27
TOLERANCE	Tolerant Species Richness tol
TROPHIC	Abundance of Cyprinid Invertivores
TROPHIC	Abundance of Native Benthic Invertivore Individuals
TROPHIC	Cyprinid Invertivore Species Richness
TROPHIC	Herbivore Species Richness
TROPHIC	Invertivore Species Richness
TROPHIC	Invertivore/Piscivore Species Richness
TROPHIC	Native Benthic Invertivore Species Richness
TROPHIC	Native Herbivore Species Richness
TROPHIC	Native Invertivore Species Richness
TROPHIC	Native Invertivore/Piscivore Species Richness

TROPHIC	Native Piscivore Species Richness
TROPHIC	Omnivore Species Richness
TROPHIC	Piscivore Species Richness
TROPHIC	Proportion of All Species that are Cyprinid Invertivores
TROPHIC	Proportion of All Species that are Herbivore
TROPHIC	Proportion of All Species that are Invertivore
TROPHIC	Proportion of All Species that are Invertivore/Piscivore
TROPHIC	Proportion of All Species that are Native Benthic Invertivores
TROPHIC	Proportion of All Species that are Native Herbivore
TROPHIC	Proportion of All Species that are Native Invertivore
TROPHIC	Proportion of All Species that are Native Invertivore/Piscivore
TROPHIC	Proportion of All Species that are Native Nontolerant Herbivore int
TROPHIC	Proportion of All Species that are Native Nontolerant Herbivore int + mod
TROPHIC	Proportion of All Species that are Native Nontolerant Invertivore int + mod
TROPHIC	Proportion of All Species that are Native Nontolerant Invertivore/Piscivore int + mod
TROPHIC	Proportion of All Species that are Native Nontolerant Piscivore int + mod
TROPHIC	Proportion of All Species that are Native Piscivore
TROPHIC	Proportion of All Species that are Native Sensitive Invertivore int
TROPHIC	Proportion of All Species that are Native Sensitive Invertivore/Piscivore int
TROPHIC	Proportion of All Species that are Native Sensitive Piscivore int
TROPHIC	Proportion of All Species that are Nontolerant Herbivore int
TROPHIC	Proportion of All Species that are Nontolerant Herbivore int + mod
TROPHIC	Proportion of All Species that are Nontolerant Invertivore int + mod
TROPHIC	Proportion of All Species that are Nontolerant Invertivore/Piscivore int + mod
TROPHIC	Proportion of All Species that are Nontolerant Piscivore int + mod
TROPHIC	Proportion of All Species that are Omnivore
TROPHIC	Proportion of All Species that are Piscivore
TROPHIC	Proportion of All Species that are Sensitive Invertivore int
TROPHIC	Proportion of All Species that are Sensitive Invertivore/Piscivore int
TROPHIC	Proportion of All Species that are Sensitive Piscivore int
TROPHIC	Proportion of Cyprinid Individuals that are Omnivore (Steedman 1988)
TROPHIC	Proportion of Individuals that are Cyprinid Invertivores
TROPHIC	Proportion of Individuals that are Herbivore
TROPHIC	Proportion of Individuals that are Invertivore
TROPHIC	Proportion of Individuals that are Invertivore/Piscivore
TROPHIC	Proportion of Individuals that are Native Benthic Invertivores
TROPHIC	Proportion of Individuals that are Native Herbivore
TROPHIC	Proportion of Individuals that are Native Invertivore
TROPHIC	Proportion of Individuals that are Native Invertivore/Piscivore
TROPHIC	Proportion of Individuals that are Native Nontolerant Herbivore int
TROPHIC	Proportion of Individuals that are Native Nontolerant Herbivore int + mod
TROPHIC	Proportion of Individuals that are Native Nontolerant Invertivore int + mod
TROPHIC	Proportion of Individuals that are Native Nontolerant Invertivore/Piscivore int + mod

TROPHIC	Proportion of Individuals that are Native Nontolerant Piscivore int + mod
TROPHIC	Proportion of Individuals that are Native Piscivore
TROPHIC	Proportion of Individuals that are Native Sensitive Invertivore int
TROPHIC	Proportion of Individuals that are Native Sensitive Invertivore/Piscivore int
TROPHIC	Proportion of Individuals that are Native Sensitive Piscivore int
TROPHIC	Proportion of Individuals that are Nontolerant Herbivore int
TROPHIC	Proportion of Individuals that are Nontolerant Herbivore int + mod
TROPHIC	Proportion of Individuals that are Nontolerant Invertivore int + mod
TROPHIC	Proportion of Individuals that are Nontolerant Invertivore/Piscivore int + mod
TROPHIC	Proportion of Individuals that are Nontolerant Piscivore int + mod
TROPHIC	Proportion of Individuals that are Omnivore
TROPHIC	Proportion of Individuals that are Piscivore
TROPHIC	Proportion of Individuals that are Sensitive Invertivore int
TROPHIC	Proportion of Individuals that are Sensitive Invertivore/Piscivore int
TROPHIC	Proportion of Individuals that are Sensitive Piscivore int

NonTolerant refers to all species that are intolerant and moderately tolerant; int refers to all species that are intolerant; mod refers to all species that are moderately tolerant under the Tol. column in Table A.1. Lotic refers to all species with an X under the Lotic column in Table A.1.

Lithophilic refers to all species that are A13, A23, A12 under the Repr. column in Table A.1.

Non-lithophilic nest guarder refers to all species that are B and B27 under the Repr. column in Table A.1. General spawner refers to all species that are A11 under the Repr. column in Table A.1. Sensitive spawner refers to all species that are A23, A24, and B27 under the Repr. column in Table A.1.

Chapter 3. Identifying environmental mechanisms regulating IBI scores of two ecoregions in South Dakota

This chapter is being prepared for submission to Ecological Indicators and was co-authored by Katie N. Bertrand, Lyntausha Kuehl, Aaron Suehring, and Nels H. Troelstrup, Jr.

Abstract

Fish and environmental data were collected from 178 wadeable perennial stream reaches in two Level III ecoregions in South Dakota to assess the habitat drivers of fish assemblage structure. Understanding the relationship of reach level and regional environmental drivers of fish communities allows for mitigation by local and regional restoration efforts to improve water quality. We assessed 22 habitat variables in three different categories (chemical, reach level, and geomorphology) by first using only habitat variables that were significantly correlated through linear regression $P < 0.05$, then removed redundant variables with Spearman rank coefficients greater than $r > 0.70$ and assessed the final environmental variables with Canonical Correspondence Analysis. The habitat variables that passed the screening were different between the two ecoregions, but within ecoregions, the remaining habitat variables varied significantly with IBI score. For the Northern Glaciated Plains ecoregion remaining habitat variables accounted for 19.7% of the variation in IBI metric scores with dissolved oxygen (DO) and phosphorous strongly correlated with CCA Axis 1 and discharge and stream width strongly correlated with CCA Axis 2. In the Northwestern Great Plains ecoregion remaining habitat variables explained 32.9% of the variation in IBI metric scores with DO, total Kjeldahl nitrogen, and mean width by depth ratio partitioning sites along CCA Axis 1 and large substrate and stream discharge partitioning sites along CCA Axis 2. Each of these local habitat variables identified as drivers of fish communities could be targeted by biologists and land managers to restore biotic integrity and improve ecosystem structure and function.

Introduction

Freshwater fish communities are vulnerable to losses of biodiversity through anthropogenic disruption of both intrinsic and extrinsic variables at multiple spatial scales (Olden and Jackson 2001). Agricultural intensification and increasing urban land use has had a profound impact on stream ecosystems and resident fish assemblages (Diana et al. 2006). Fish assemblages are responsive to land use and habitat conversion and the effects can be assessed by species-habitat interactions (Richards et al 1996; Meador and Goldstein 2003; Wang et al. 2003; Sindt et al. 2012). The combination of abiotic and biotic effects operating at multiple special scales makes prediction of fish assemblage structure difficult (Poff 1997; Marsh-Matthews and Matthews 2000; Jackson et al. 2001; Sindt et al. 2012). However, any model should assess species-habitat associations at multiple scales (Leftwhich et al. 1997; Rich et al. 2003; Pont et al. 2005; Hoeinghaus et al. 2007; Sindt et al 2012). Fish assemblage variation has been explained at coarse scale (Marsh-Matthews and Matthews 2000) and fine scale (Gorman and Karr 1978; Schlosser 1982; Hubert and Rahel 1989; Lobb and Orth 1991), but manipulating fine scale habitat variables to achieve management objectives is more tractable than trying to affect coarse scale (i.e., watershed) change (Sindt et al. 2012). Examples of fine scale management actions include increasing riparian buffer width to reduce the input of sediment, planting trees within the riparian area to shield buffer temperature increases, or removing barriers to increase connectivity.

The Index of Biotic Integrity (IBI) is a widely accepted tool for assessing water quality based on characteristics of the biotic community (Yoder and Rankin, 1998; Simon, 1999; Quist, 2001). The benefit of the IBI is its ability to identify stressors other than point source chemical pollutants (Karr 1981, Karr 1987, Fausch 1990). With a more a holistic goal in mind the paradigm has changed from the approach of stream water quality assessment using pollution standards to the use of fish (or invertebrate) community associations and analysis. The disadvantage of the IBI is that it does not make the final critical connection that links the species (biologic attribute, termed metrics) to actionable changes to improve water quality. Meaning, there is still a gap in our knowledge as to why fish occur in a particular reach. Local environmental conditions may be better predictors of fish communities (Lammert and Allen 1999; Meador and Glostein 2003). Those modifications could systematically improve specific habitats that are critical for select species, resulting in an increased IBI score, ultimately leading to improved stream health.

In South Dakota (SD) large amounts of native prairie were historically and are currently being converted to row crop agriculture or pasture land with increased grazing pressure. In a study conducted by Wright and Wimberly (2012) in the Western Corn Belt (North and South Dakota, Minnesota, Iowa and Nebraska), about 530,000 ha were converted from grassland to cropland, with an annual rate of conversion at about 1.0-5.4 %. This pressure in SD was most intense from 2006-2011 in the eastern half of the state in the Northern Glaciated Plains (NGP). Increased conversion of native prairie to agriculture can lead to increases in sedimentation, changes in stream morphology, nutrient enrichment, and increased flooding or drying, all forms of degradation (Omernick et al. 1981; Smart et al. 1981; Osborne and Wiley 1988; Karr and Chu, 2000; Malmqvist and Rundle, 2002). The James and Des Moines lobes of the Wisconsinian Glacier 150,000 shaped the NGP million years ago (Flint, 1955). Glacial retreat left behind many wetlands shallow lakes and fertile soils. Glaciation also shaped the river and stream systems in SD. The major river systems in the NGP (James River, Vermillion River, and Big Sioux River) run north to south (Flint, 1955). This is in contrast to the major river systems of the Northwestern Great Plains (NWGP) in western SD (Grand River, Moreau River, Cheyenne River, and White River) where they run from the west to east. Western SD is has less fertile soils and limited precipitation and as a result is dominated by cattle grazing (Bryce et al. 1998), with row crop agriculture intensifying as drought resistant crops become more prevalent (U.S. Environmental Protection Agency, 1996).

Assessments of biotic integrity are critically important in classifying and monitoring streams in South Dakota. Index of Biotic Integrity scores are derived based on the fish present at a stream at the time of its assessment. To improve IBI scores at a given site, positive metrics must increase and/or negative metrics must decrease. This is a simple theory, but in reality, the fish must be able to move freely throughout the region and meet their basic life requirements, in order to establish in a “restored” reach. The objective of our study was to identify and assess the habitat variables from two Level (LV) III ecoregions (NGP and NWGP) in South Dakota that are most influential on fish communities. Environmental Monitoring and Assessment Program (EMAP) protocols, plus additional spatial scale habitat variables. This comparison was conducted by using multivariate statistical modeling programs. Multiple analyses were used to assess the most influential habitat variables affecting species distributions in the NGP ecoregion and NWGP ecoregion. Both Ecoregions have recently developed specific IBIs (Krause et al. 2013 and Kaiser et al. Unpublished). The habitat variables more associated with positive metrics would be an environmental variable that potentially has a positive impact on water quality. Alternatively any negative habitat metric associated with habitat variables could show environmental variables which have degraded enough to negatively affect stream quality. We hypothesized that by assessing the drivers of community structure we should be able to forecast potential fish habitat related stressors that affect fish assemblage distributions and, either prevent degradation to critical fish community habitats or identify actionable habitat features for mitigation.

Methods

Sampling Area

Sites were located on wadeable, perennial streams in two of South Dakota's LV III ecoregions: the NGP ecoregion and the NWGP ecoregion. The NGP originates in south eastern SD and extends north through South Dakota and North Dakota into the provinces of Saskatchewan and Manitoba in Canada. It covers approximately one-third of the state situated on the eastern border of SD. The NGP ecoregion is comprised of 15 LIV ecoregions, eight of which are in SD. Of the eight LIV ecoregions in SD, six were sampled for this study, the Glacial Lakes Deltas and Tewaukon Dead Ice Moraine ecoregions were not sampled as a result of their disproportionately smaller area relative to the other six. The NGP ecoregion's climate is subhumid with 43 to 56 cm of precipitation falling annually and native vegetation made up of mixed and tallgrass prairie species (Bryce et al. 1998).

The NWGP ecoregion extends from the base of the Rocky Mountains in Montana east into the southwestern corner of North Dakota down through most of western South

Dakota and west into the northeast corner of Wyoming. In South Dakota the NWGP ecoregion is located entirely west of the Missouri River and accounts for approximately one-half of SD's surficial drainage area (Bryce et al. 1998). The ecoregion includes eleven LIV ecoregions, of which eight were sampled (Fig. 1) and one, Missouri Badlands that resides entirely outside of the state. The Forested Buttes and the Dense Clay Prairie LIV ecoregions were eliminated, the former being high gradient rainwater runoff gullies and the later lacking enough perennial wadeable streams to compose a statistical average. Climate within this ecoregion is semiarid and natural vegetation is primarily mixed and short grass prairie species. Soils within this ecoregion are derived from shale, siltstone and sandstone (Bryce et al. 1998). Cattle grazing dominates the landscape, but spring wheat and alfalfa are common crops, large areas of native grasslands are present (Bryce et al. 1998; Chapman et al. 2001; U.S. Environmental Protection Agency 1996). Agriculture is limited by erratic precipitation. Mean annual precipitation ranges from 25 to 51 cm (Chapman et al. 2001).

We sampled 178 sites, 84 of which were sampled twice. In the NGP ecoregion, 58 sites were sampled, 28 of which were sampled twice, and in the NWGP ecoregion 65 sites were sampled, 56 of which were sampled twice. In the NGP ecoregion 60 sites were selected based on predetermined disturbance level (Fig. 3-1). The eight highest ranking sites were selected from among sites in the lower 5th percentile for water quality violations, and the eight lowest ranking sites were selected from among the upper 5th percentile of water quality violations. Fifteen random sites also were sampled, with four sites removed from analysis because they lacked fish (Krause et al, 2013).

In the NWGP ecoregion stream reaches were selected at random from a target population of over 7,000 stream segments throughout South Dakota's portion of the NWGP ecoregion. One hundred twenty sites were stratified by LIV ecoregion. The number of sites in each LIV ecoregion was proportional to the number of river kilometers in that ecoregion. Sites located immediately below an impoundment or natural basin (within a 5 km buffer) were excluded. If sites were inaccessible or we were unable to get permission to sample a site, another was chosen from within the same LIV ecoregion at random to replace it. This provided us with a probability-based random sample of wadeable stream sites, allowing for characterization of stream condition within each LIV ecoregion and across the NWGP as a whole.

Assessments of a single site comprised all of the following samples: water chemistry (Table 3-1), fish, and physical habitat, each was assessed once during each growing season from June to August in 2010 and 2011 for the NGP ecoregion and 2014 and 2015 for the NWGP ecoregion. Samples were collected following Standard Operating Procedures (SOP) for Field Samplers, Volume II, Biological and Habitat Sampling (SD DENR, 2005). All sites were sampled below bankfull conditions. Reach length was acquired by measuring the wetted width at 10 locations within the target stream segment. Those 10 measurements were averaged to estimate the preliminary mean stream width (PMSW). If the PMSW is less than or equal to 10 m, transects were spaced three PMSWs apart. If the PMSW is greater than 10 m, transects were spaced two PMSWs apart. The total number of transects was 11 at each site. We instituted a minimum of 100 m and a max of 300 m reach length as a bench mark for both very narrow streams and wide wadeable streams.

Fish Data

Fish were collected with a seine or by backpack electrofishing, depending on the stream channel conditions, with block nets set at upper and lower transects to prevent fish escapement. If the stream channel contained significant obstructions, such as aquatic vegetation or large rocks, electrofishing was employed to sample that reach, otherwise, seines were used. Every effort was taken to collect fish observed from all habitat types available within the sampled reach. In very small streams (<2 m wide) it was possible to sample most of the available habitat, but in larger streams, we meandered in an upstream direction between habitat types. Fish survey results were recorded, including the specimen identification to species, length measurement, and counts were generally made in the field as samples are drawn from field gear. However, some species and small specimens required transport back to the laboratory for closer inspection. Fish less than 25 mm in total length were not counted as part of the catch. Voucher specimens of each fish species were retained for quality control and assurance purposes and deposition into the Willis Fisheries Museum at SDSU. For fish that were identified with certainty to species level, several individuals of each species were preserved in 10% formalin solution. All fish that could not be identified to the species level were preserved in a separate container in a 10% formalin solution.

Water Chemistry

Variables linked to water quality criteria in support of beneficial stream uses in South Dakota were measured from each of the 65 target reaches. Water quality grab samples and multiparameter probe measurements were collected upstream at transect 11 within each sampled stream reach. During the collection of water quality samples, instantaneous discharge measurements was taken at transect 1, 6, and 11. All water quality samples will be collected using the methods outlined in Standard Operating Procedures for Field Samples Volume 1 Tributary and In-Lake Sampling Techniques (SD DENR 2005).

Physical Habitat

Detailed physical habitat measurements were taken from each site following collection of water chemistries and biological samples (SD DENR Water Resources Assistance Program, 2005). Habitat data was collected from the entire sample reach and eleven equally spaced transects placed at equidistant locations along the reach. On either end of a transect the riparian land use, dominant vegetation type, animal vegetation use, dominant bank substrate, and bank slumping (presence/absence) were recorded. At eight locations across each transect bed substrate measurements were collected. Several measurements of the channel cross-section were collected to estimate stream width, depth, channel bottom and top width, water depth, channel slope, bank length, bank angle, bank height, bankfull width, bankfull depth, and width:depth ratio. Also the length of the banks that were vegetated, erosional or depositional, as well as horizontal length of over-hanging vegetation and undercut banks extending over the stream channel bed were measured. Canopy cover was also collected from six stations at each transect using a densiometer. Finally, the number of large woody debris (LWD) were tallied for the entire reach. Length and diameter of all pieces of LWD (> 5 cm diameter) were measured to calculate the volume of LWD within the reach.

Analysis

The two LV III ecoregions were initially assessed using a Discriminant Function Analysis (DFA). This test was used to assess the differences between the LV III and IV ecoregions. Data were analyzed following steps similar to those in D'Ambrosio et al. (2007). Although we initially considered assessing these data sets at just the fine scale, Hoeinghaus et al. (2007) found using canonical correspondence analysis (CCA) that more variation was explained by combining both local and regional scales. The 22 habitat variables were grouped into three different categories: water chemistry (eight variables), geomorphology (five variables), and reach level (nine variables). All habitat variables were regressed against regional IBI scores created specifically for each LV III ecoregion (Krause 2013, C.K. Kaiser Unpublished). If there was a significant relationship ($P < 0.05$) to IBI score, the habitat variable was retained for further analysis. Additional independent variables were reduced by removing highly correlated variables. A habitat variable was considered highly correlated if there was a Spearman rank coefficient $r > 0.70$ (Table 3-2). Finally CCA analysis was used to compare the fish assemblage characteristics (IBI metrics) to the remaining habitat variables for each LV IV ecoregion. Within a CCA diagram both the community composition and habitat variables are plotted to best represent the variation within the community and the relations between species and the habitat variables (Jongman et al. 1995) in this instance our "community" is the metrics generated from the IBI (D'Ambrosio et al. 2009). This analysis allows us to use the habitat variables, which are plotted within the ordination as vectors. With the length of the vector showing the influence of habitat variables on the fish species.

Results

Our analysis included 99,600 fish, representing 48 species and ten families. In the NGP ecoregion 60,373 fish were sampled, and the remaining 39,227 fish were sampled from the NWGP ecoregion. In the NGP ecoregion, 82% of the total catch was comprised of eight species and three families. The most abundant family was Cyprinidae (70%) and was represented by fathead minnow (*Pimephales promelas*), common shiner (*Luxilus cornutus*), sand shiner (*Notropis stramineus*), creek chub (*Semotilus atromaculatus*), brassy minnow (*Hybognathus haninsoni*), and emerald shiner (*Notropis atherinoides*). The two additional families were Ictaluridae (7%) and Percidae (5%) represented by one species each black bullhead (*Ameiurus melas*) and Johnny darter (*Etheostoma nigrum*), respectively. In the NWGP ecoregion 80% of the total catch was comprised of four species and two families. The most abundant family was Cyprinidae (77%) and was represented by fathead minnow (*Pimephales promelas*), red shiner (*Cyprinella lutrensis*), sand shiner (*Notropis stramineus*), and creek chub (*Semotilus atromaculatus*). The one additional family was Clupeidae (3%) represented by gizzard shad (*Dorosoma cepedianum*).

Analysis of the habitat using DFA showed 40% correct reclassification rate for LV IV ecoregions and 83% correct reclassification rate for LV III ecoregions. Discriminant Function Analysis of the fish resulted in 31% correct reclassification rate for LV IV ecoregions and 79% correct reclassification rate for LV III ecoregions.

Within the two ecoregions, habitat was notably variable; DO varied from a low 0.24 mg/L^{-1} to a high of 13.97 mg/L^{-1} with an average of 5.78 ± 0.37 mg/L^{-1} in the NGP ecoregion. In the NWGP ecoregion DO varied from 0.24 mg/L^{-1} to 15.43 mg/L^{-1} with an average of 8.1 ± 0.2 mg/L^{-1}. The percentage of the reach that was large substrate (sand and larger) varied from 0% to 73% with an average of 31.61 ± 3.13% in the NGP ecoregion and 0% to 100% with an average of 54.67 ± 2.89% in the NWGP ecoregion. Geomorphology was more similar, average stream width was within about one meter. The NGP ecoregion had an average stream width of 5.81 ± 0.37 m and the NWGP ecoregion averaged 4.78 ± 0.28 m. Streams in the NWGP ecoregion were found, on average, to be deeper than the streams in the NGP ecoregion. Width to depth ratios for the NGP ecoregion were 12.06 ± 0.87 and 8.86 ± 0.59 in the NWGP ecoregion.

A total of 22 habitat variables were assessed for correlation with IBI score and only ten habitat variables were significantly ($P < .05$) related to IBI score in the NGP ecoregion and seven in the NWGP ecoregion. The variables found to be significant were assessed for correlation (Spearman rank correlation) within the three groups. A habitat variable was removed if it had a Spearman rank correlation $r > 0.70$, four habitat variables from each ecoregion were removed from analysis with this test.

Index of Biotic Integrity metrics were used to generate a matrix of species data for each site in the CCA. In the NGP ecoregion, Centrarchidae plus Largemouth Bass (LMB) Richness, Proportion of Fish Species that are Alien, and Total Tolerant Species Richness are negative metrics. Positive metrics are Proportion of Individuals that are Native Cool Water Species, Proportion of All Species that are Lithophilic Spawners, and the Proportion of Individuals that are Native Nontolerant Invertivores. In the NWGP ecoregion the positive metrics were Cyprinid Invertivore Species Richness, Proportion of All Species that are Native Lithophilic Spawners, Proportion of Individuals that are Native Large River Migrants, and Proportion of Individuals that are Longnose Dace

(*Rhinichthys cataractae*). The negative metrics in the NWGP ecoregion were the Proportion of All Species that are Tolerant and Abundance of Alien Fish.

The total variation in fish assemblage structure (IBI metric scores) explained by the habitat variables was 19.7%, with axis 1 explaining 12.0% of the variation and axis 2 explaining 7.7% in the NGP ecoregion (Fig. 3-2). In the NWGP ecoregion the total variation of the IBI explained by the habitat variables was 32.9%, with axis 1 explaining 20.3% of the variation and axis 2 explaining 12.6% (Fig. 3-3).

The positive NGP ecoregion IBI metrics, Proportion of All Species that are Lithophilic Spawners and Proportion of Individuals that are Native Cool Water Species increased as mean bank angle, dissolved oxygen, total canopy cover and large substrate inclined. The other two negative metrics, Total Tolerant Species Richness metric and Proportion of Fish Species that are Alien metric were closer to center on the first axis but the Proportion of Fish Species that are Alien metric increased with increased discharge and mean stream width, while Total Tolerant Species Richness metrics increased with declining discharge and mean stream width. The negative metric, the Centrarchidae plus LMB richness metric showed a high correlation to increases in phosphorus and mean width to depth along the first axis. Species richness of Centrarchidae and LMB also increased as mean bank angle, dissolved oxygen, total canopy cover, and large substrate declined. The Proportion of Individuals that are Native Nontolerant Invertivore metric showed no real association to any of the vectors in the CCA.

In the CCA plot for NWGP ecoregion, the two negative metrics: Proportion of All Species that are Tolerant and Abundance of Alien Fish were associated with increases in detritus, total Kjeldahl nitrogen, and the increase in the percent of the banks that were vegetated. These two metrics were also influenced slightly by increases in conductivity. Along the first axis these metrics increased when dissolved oxygen and the mean width to depth ratio decreased. Three of the four positive metrics Cyprinid Invertivore Species Richness, Proportion of Individuals that are Native Large River Migrants, and Proportion of Individuals that are LOD increased with increases in dissolved oxygen and mean stream width by depth but all respond differently on the second axis. The Proportion of Individuals that are LOD is strongly influenced by increases in mean discharge and the subsequent increase in substrate size. The Proportion of Individuals that are Native Large River Migrants increased predominantly with mean stream width, but appears to be influenced by increases in turbidity and the amount of the bank that is eroded. Cyprinid Invertivore Species Richness encompassed fish from both of the previous metrics and showed little response to the second axis (Fig. 3-3). These three positive metrics are plotted on the right side of the CCA plot, and appeared to respond to decreases of detritus, total Kjeldahl nitrogen, and an increase in the percent of the banks that were vegetated.

Discussion

As evident by the high correct reclassification rate of the DFA when comparing the two LV III ecoregions, these two ecoregions represent two distinct environmental features. When we compared all LV IV ecoregions the reclassification rate dropped to about 40%. This shows that at finer resolution there were fewer differences in the habitat. Similarly, the NGP and NWGP fish assemblages were distinct. Longitude appears to be the first coarse filter of fish communities in SD. The communities in western SD are first historic remnants, after the glaciation of eastern SD the two ecoregions became separated and fish communities east of the Missouri River were extirpated. The current assemblage of eastern SD is representative of the fish that were able to recolonize after the glacier receded. These populations were further divided by environmental conditions. The western side of the state receives less annual precipitation than the east and due in part to geologic structure the streams in western SD are more prone to seasonal drying. These two aquatic assemblages were further removed from each other in the 1940's when multiple dams along the Missouri River began construction.

Notably, the Prairie Coteau Escarpment LV IV ecoregion in the NGP ecoregion and Sagebrush Steppe LV IV ecoregion in the NWGP ecoregion had the highest average IBI scores within their respective LV III ecoregion. The results of the DFA show that the habitats in two distinct ecoregions in SD could be affecting IBI scores. The LV IV NGP ecoregions showed more diversity in the fish assemblage than the NWGP LVIV ecoregion. These different fish assemblages showed no relationship to IBI scores. Across the larger level III ecoregion there was high variability in the physical habitat attributes. We know that fish respond to a gradient of environmental filters to ultimately result in capture at a local reach (Gorman and Karr, 1978; Waters, 1995; Hoeinghaus et al. 2007) and that local habitat values are the result of landscape conditions that have been shaped by multiple higher order environmental filters and processes (Hocutt and Wiley, 1986; Fisher and Paukert 2008).

These results indicate that in the NGP ecoregion, streams of higher quality, with regards to IBI scores, are more associated with increased canopy cover, dissolved oxygen, and large substrate, all of which would increase dissolved oxygen. And alternatively as mean width to depth and total phosphorus increases the water quality becomes poorer. Both are signs of increased agriculture activity (Richards et al. 1996).

In the Midwest, agriculture (row crop and cattle grazing) can contribute over 90% of total nitrogen and phosphorous transport in a watershed (Becher et al. 2000). Increases in substrate size are often associated with increases in biotic integrity and biodiversity allowing for spawning and hiding areas (Sindt et al. 2012). One of the species represented by this group was the Flathead Chub (*Platygobio gracilis*) which typically inhabits turbid streams and rivers (Rahel and Thel, 2004). Anthropogenic disturbance can affect geomorphology which can effect assemblage structure (Infante et al. 2006). Intensive cattle grazing can cause increases in width and decreases in depth, which can lead to increases in temperature (Magilligan and McDowell 1997). Trampling by cattle can increase the sloughing of stream banks which can lead to increases in sediment input (Magilligan and McDowell 1997, Armour et al. 1991). These disturbances decrease available habitat for fish and limit recruitment and growth potential.

The Whittier process of IBI metric selection takes steps to remove the effect of natural gradients from influencing the final metric selection. This process interacts with stream order (i.e., Vannote et al. 1980). As streams become larger more habitat is available and as long as anthropogenic disturbance is relatively low, higher diversity is expected at these sites (Marsh-Matthews and Matthews 2000). This is reflected in our results of the assessment of geomorphology, but with two unique results. In Eastern SD, IBI scores decreased with increasing stream width and the opposite was true in the NWGP ecoregion. In the NGP, increasing stream width comes at the cost of incorporating much greater anthropogenic disturbance resulting in lower IBI scores.

NWGP showed a positive correlation to IBI score and stream width, part of the Whittier process of metric selection is to eliminate natural gradients, no single IBI metric showed a positive correlation to stream width. But as a result of the metrics being what they were, the combination of metrics showed a correlation in the final score with stream width. East river was the opposite, maybe a result of lower width to depth ratio. As streams east river become wider they also become shallower, possible as a result of sedimentation corresponding decreases in dissolved oxygen. Thus fish necessary to achieve higher scores in each of the metrics are not collected.

Increases in nitrogen and phosphorus can result in eutrophication, losses of diversity and decreases in dissolved oxygen, which can lead to fish kills (Carpenter et al. 1998). Increases in intensive agriculture activity have been shown to increase stream nitrogen and phosphorus (Bennett et al. 2001, Bernot et al. 2006). Intensification of agriculture land use can also effect channel morphology, Gucker et al. (2009) found that streams in areas with increased agriculture activity were shallower and more homogenous. These habitat variables (phosphorus, DO, and width/depth) are leading factors describing negative metrics in the NGP, and are easily mitigated at a single site, but can be complex to mitigate watershed-wide. Riparian buffers and fences could be implemented to reduce the effects of sedimentation, grazing cattle, and chemical runoff. Borin et al. (2005) found that buffer strips decrease the amount of phosphorus that enters streams, specifically the sediment bound phosphorus, where buffer strips would also lower the amount of sediment entering the streams.

Reach level restoration efforts are critically important to the ability of sensitive species to recolonize. Each reach’s habitat characteristics are a product of not only the land use surrounding the stream at a given point, but they also reflect the legacy of the degradation upstream (Poff 1997; Marsh-Matthews and Matthews 2000; Jackson et al. 2001; Sindt et al. 2012). Without basin wide best management practices, fish that drive IBI scores will not have the ability to move or survive at specific points.

One aspect of the NWGP that we did not assess was the disturbance regime. Fish communities are formed not only by life history traits but also by spatial patterns and connectivity in an area of frequent disturbance such as prairie streams (Schlosser 1982; Dodds et al. 2004). Variables such as wetting and drying cycles, connectivity, and distance from mainstem rivers (recolonization) should be assessed. Areas with large segments of streams which are prone to low flows, have reaches that day and become physical barriers. Another consideration is the relatively low anthropogenic disturbance in western South Dakota. The conditions represented in our study lead to the concept that these prairie streams represent a pristine environment where, for the fish species that can tolerate the harsh environmental variability, are able to persist without extirpation due to human induced stressors. Continued monitoring within this environment is paramount to track disturbance, whether disturbance comes in the form of climate change or anthropogenic disturbance.

Literature Cited

Armour, C. L., D. A. Duff, and W. Elmore, 1991. The Effects of Livestock Grazing on Riparian and Stream Ecosystems (American Fisheries Society position statement). Fisheries 16:7-11.

Becher, K. D., D. J. Schnoebelen, and K. K. B. Akers. 2000. Nutrients discharged to the Mississippi River from eastern Iowa watersheds, 1996-1997. Journal of American Water Resources Association 36:161-173

Bennett, E. M. , S. R. Carpenter, and N. F. Caraco. 2001. Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective: Increasing accumulation of phosphorus in soil threatens rivers, lakes, and coastal oceans with eutrophication. BioScience 51 (3): 227-234.

Bernot, M. J., J. L. Tank, T. V. Royer, and M. B. David. 2006. Nutrient uptake in streams draining agriculture catchment of the midwest United States. Freshwater Biology. 51:499-509

Borin, M., M. Vianello, F. Morari, and G. Zanin. 2005. Effectiveness of buffer strips in removing pollutants in runoff from a cultivated field in North-East Italy. Agriculture, Ecosystems and Environment 105:101–114

Bryce, S.A., J.M. Omernik, D.E. Pater, M. Ulmer, J. Schaar, J. Freeouf, R. Johnson, P. Kuck, and S.H.Azevedo. 1998. Ecoregions of North Dakota and South Dakota. (Map poster). U.S. Geological Survey,Reston, VA.

Chapman, S. S., J. M. Omernik, J. A. Freeouf, D. G. Huggins, J. R. McCauley, C. C. Freeman, G. Steinauer, R. T. Angelo, and R. L. Schlepp. 2001. Ecoregions of Nebraska and Kansas (color poster with map, descriptive text, summary tables, and photographs): Reston, Virginia, U.S. Geological Survey (map scale 1:1,950,000).

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface water with phosphorus and nitrogen. Ecological Applications 8(3):559-568

D'Ambrosio, J.L., L. R. Williams, J. D. Witter, A. Ward. 2009. Effects of Geomorphology, habitat, and spatial location on fish assemblages in a watershed in Ohio, USA. Environmental Monitoring and Assessment 148:325-341

Diana, M., J. D. Allan, and D. Infante. 2006. The Influence of Physical Habitat and Land Use on Stream Fish Assemblages in Southeastern Michigan. American Fisheries Society Symposium 48:359-374

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, W. J. Matthews. 2004. Life on the edge: the ecology of Great Plains prairie streams. BioScience 54(3):205-216

Fausch, K. D., J. Lyons, J. R. Karr, and P. L. Angermeier. 1990. Fish communities as indicators of environmental degradation. American Fisheries Society Symposium, 8:123-144.

Fischer, J. R. and C. P. Paukert. 2008. Habitat relationships with fish assemblages in minimally disturbed Great Plains regions. Ecology of Freshwater Fish 17:597-609.

Flint, R. F. 1955. Pleistocene geology of Eastern South Dakota. Washington, D.C., U.S. Geological Survey, U.S. Government Printing Office, Professional Paper 262, 173 p.

Gorman, O.T., and J. R. Karr 1978. Habitat Structure and Stream Fish Communities. Ecology 59:507–515.

Gucker, B., I. G. Boechat, and A. Giani. 2009. Impacts of agriculture land use on ecosystem structure and whole-stream metabolism of tropical Cerrado streams. Freshwater Biology 54:2069-2085

H ocutt, C. H. and E. O. Wiley. 1986. The zoogeography of North American freshwater fishes. John Wiley & Sons, New York. 866 p.

Hoeinghaus D. J., K. O. Winemiller, and J. S. Birnbaum. 2007. Local and regional determinants of stream fish assemblage structure: inferences based on taxonomic vs. functional groups. Journal of Biogeography 34: 324-338.

Hubert, W. A. and F. J. Rahel. 1989. Relations of physical habitat to abundance of four nongame fishes in high-plains streams; a test of habitat suitability index models. North American Journal of Fisheries Management 9:332-340.

Infante, D. M., M. J. Wiley, and P. W. Seelbach. (2006). Relationships among channel shape, catchment characteristics, and fish in lower Michigan streams. American Fisheries Society Symposium 2006(48), 339-357.

Jackson, D. A., P. R. Peres-Neto, and J. D. Olden. 2001. What controls who is where in freshwater fish communities-the roles of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58:157–170.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities, Fisheries, 6(6), 21-27.

Karr, J. R. 1987. Biological monitoring and environmental assessment: a conceptual framework. Environmental Management 11:249-256.

Karr, J. R. and Chu, E. W., 2000. Sustaining living rivers. Hydrobiologia 422, 1-14.

Krause, J. R., K. N. Bertrand, A. Kafle, and N.H. Troelstrup. 2013. A fish index of biotic integrity for South Dakota's Northern Glaciated Plains Ecoregion. Ecological Indicators, 34, 313-322.

Lammert, M., and J. D. Allen. 1999. Assessiond the biotic integrity of streams: effects of scale in measuring the influence of land use/ cover and habitat structure on fish and macroinvertebrates. Environmental Management 23::257-270.

Leftwich, K. N., P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing behavior and transferability of habitat models for a benthic stream fish. Transactions of the American Fisheries Society 126:725-734.

Lobb, M. D. III, and D. J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. Transactions of the American Fisheries Society 120:65–78.

Magilligan, F. J., and P. F. McDowell. 1997. Stream channel adjustments following elimination of cattle grazing. Journal of the American Water Resources Association 33:867–878.

Malmqvist, B., S. Rundle, 2002. Threats to the running water ecosystems of the world. Environmental Conservation 29:134-153.

Marsh-Matthews, E., and W. J. Matthews. 2000. Geographic, terrestrial and aquatic factors: which most influence the structure of stream fish assemblages in the midwestern United States. Ecology of Freshwater Fish 9:9–21.

Meador, M. R., and R. M. Goldstein. 2003. Assessing water quality at large geographic scales: Relations among land use, water physicochemistry, riparian condition, and fish community structure. Environmental Management 31(4):504-517

Olden, J. D., and D. A. Jackson. 2001. Fish–habitat relationships in lakes: gaining predictive and explanatory insight by using artificial neural networks. Transactions of the American Fisheries Society 130(5):878-897.

Omernik, J. M., A. L. Gallant, 1988. Ecoregions of the upper midwest states. U.S. Environmental Protection Agency, Corvallis, OR.

Omernik, J. M., A. R. Abernathy, and L. M. Hale 1981. Stream nutrient levels and proximity of agriculture and forest land to streams: some relationships. Journal of soil and Water Conservation 36:227-231

Osborne, L. L., and M. J. Wiley. 1988. Empirical relationships between land use/ land cover and stream water quality in an agricultural watershed. Journal of Environmental Management 26:9-27

Poff, N. L., J. D. Allen, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47(11):769-784.

Quist, M. C., C. S. Guy. 2001. Growth and mortality of praire stream fishes: relationswith fish community and instream habitat characteristics. Ecology of Freshwater Fish 10:88-96

Rahel, F. J. and L. A. Thel. (2004, July 22). Flathead Chub (Platygobio gracilis): a technical conservation assessment. http://www.fs.fed.us/r2/projects/scp/

assessments/flatheadchub.pdf, Last accessed Feb. 19, 2017. USDA Forest Service, Rocky Mountain Region.

Rich, C. F., T. E. McMahon, B. E. Rieman, and W. L. Thompson. 2003. Small habitat, watershed, and biotic features associated with bull trout occurrence in Montana streams. Transactions of the American Fisheries Society 132:1053–1064.

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitat and biota. Canadian Journal of Aquatic science. 53(suppl. 1):295-311.

Schlosser, I. J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52:395–414.

Simon, T. P., 1999. Biological integrity and use of ecological health concepts for application to water resource characterization, in: Simon, T.P. (Ed.), Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL, pp. 97-121.

Sindt, A. R., M. C. Quist, and C. L. Pierce. 2012. Habitat associations of Fish species of greatest conservation need at multiple spatial scales in wadeable Iowa streams. North American Journal of Fisheries Management 32:1046-1061.

Smart, M. M., T. W. Barney, and J. R. Jones. 1981. Watershed impact on stream water quality: a technique for regional assessment. Journal of Soil and Water Conservation 36:297-300

U.S. Environmental Protection Agency, 1996. Level III ecoregions of the continental United States (revision of Omernik, 1987): Corvallis, Oregon, U.S. Environmental Protection Agency - National Health and Environmental Effects Research Laboratory Map M-1, various scales

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian journal of fisheries and aquatic sciences, 37(1):130-137.

Wang, L., J. Lyons, P. Rasmussen, P. Seelbach, T. Simon, M. Wiley, P. Kanehl, E. Baker, S. Niemela, and P.M. Stewart. 2003.Watershed, reach, and riparian influence on stream fish assemblages in the northern lakes and forest ecoregion, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 60:491–505.

Waters, T. F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society, Bethesda, MD. 251 p.

Wright, C. K. and M. C. Wimberly. 2013. Recent land use change in the western Corn Belt threatens grasslands and wet lands. PNAS 110 (10) 4134-4139

Yoder, C .O. and E. T. Rankin. 1998. The role of biological indicators ina state water quality management process. Environmental Monitoring and Assessment 51:61-88

Tables

Table 3-1. Water quality parameters collected at random and targeted wadeable stream sites.

Parameter	Container	Preserved	Filtered	Lab
Tot Alkalinity	A Bottle (1 Liter)	None	N	SDSU
Tot Solids	A Bottle (1 Liter)	None	N	SDSU
Tot Suspended Solids	A Bottle (1 Liter)	None	N	SDSU
Tot Dissolved Solids	A Bottle (1 Liter)	None	N	SDSU
Tot Ammonia	B Bottle (1 Liter)	Sulfuric	N	DOH
Tot Nitrate	B Bottle (1 Liter)	Sulfuric	N	DOH
Tot Kjeldahl Nitrogen	B Bottle (1 Liter)	Sulfuric	N	DOH
Tot Phosphorus	B Bottle (1 Liter)	Sulfuric	N	DOH
Diss Na	C Bottle (1 Liter)	Nitric	Y	DOH
Diss Si	C Bottle (1 Liter)	Nitric	Y	DOH
Diss Ca	C Bottle (1 Liter)	Nitric	Y	DOH
Diss Mg	C Bottle (1 Liter)	Nitric	Y	DOH
Diss Sulfate	D Bottle (1 Liter)	None	Y	DOH
Diss Cl	D Bottle (1 Liter)	None	Y	DOH
Diss Fl	D Bottle (1 Liter)	None	Y	DOH
Sol Reactive	D Bottle (1 Liter)	None	Y	DOH
Phosphorus	E Bottle (DOH	None	Y	DOH
E. coli	Bottle)	---	---	SDSU
Dissolved Oxygen	Multiparameter Probe	---	---	SDSU
Conductance	Multiparameter Probe	---	---	SDSU
pH	Multiparameter Probe	---	---	SDSU
Water Temperature	Multiparameter Probe			

Table 3-2. – Optimal metrics were selected through a filtering process using range, signal-to-noise ratios, responsiveness to disturbance, and redundancy tests until there was one metric remaining in each class and associated P- values.

		NGP	NWGP
Metric	Level	Pvalue	Pvalue
Alkalinity	Chemistry	0.169	0.655
Conductivity	Chemistry	0.18	0.0162*
DO	Chemistry	0.0085*	0.00329*
Ph	Chemistry	0.507	0.241
Phosphorus	Chemistry	0.0107*	0.9
TDS	Chemistry	0.231	0.0903
TKN	Chemistry	0.0239**	0.00203*
TSS	Chemistry	0.608	0.137
Mean bankful width	Geomorphology	0.00962**	0.0193**
Mean entrenchment width	Geomorphology	0.4891	0.865
Mean flood prone width	Geomorphology	0.0104**	0.0154**
Mean stream width	Geomorphology	0.00926*	0.00621*
Mean width by depth	Geomorphology	0.000802*	0.000879*
% Bank eroded	Reach	0.719	0.0321*
% Bank Vegetated	Reach	0.243	0.00443*
Detritus	Reach	0.0266**	0.000162*
Fine substrate	Reach	0.055	7.26E-08**
Large substrate	Reach	0.00364*	2.78E-11*
Mean bank angle	Reach	0.00572*	0.207
Mean discharge	Reach	0.0233*	0.0193*
Total canopy cover	Reach	0.0000021*	0.559
turbidity	Reach	0.764	0.00228*

* significant ** was significant but was highly correlated and removed from further analysis.

Figures

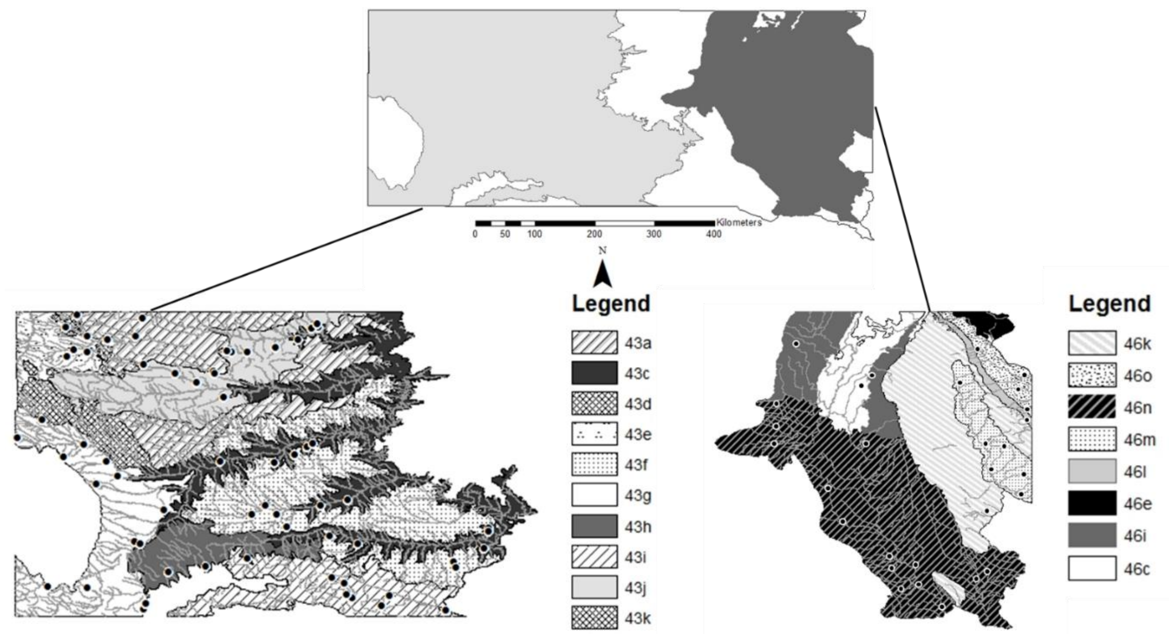

Fig. 3-1. Level IV ecoregions with their respective Level III Northern Glaciated Plains (NGP) (dark grey) Ecoregion and Northwestern Great Plains (NWGP) (light grey) Ecoregion with the state of South Dakota. Location of 95 study stream reaches samples in NGP and NWGP. Grey lines indicate perennial wadeable streams.

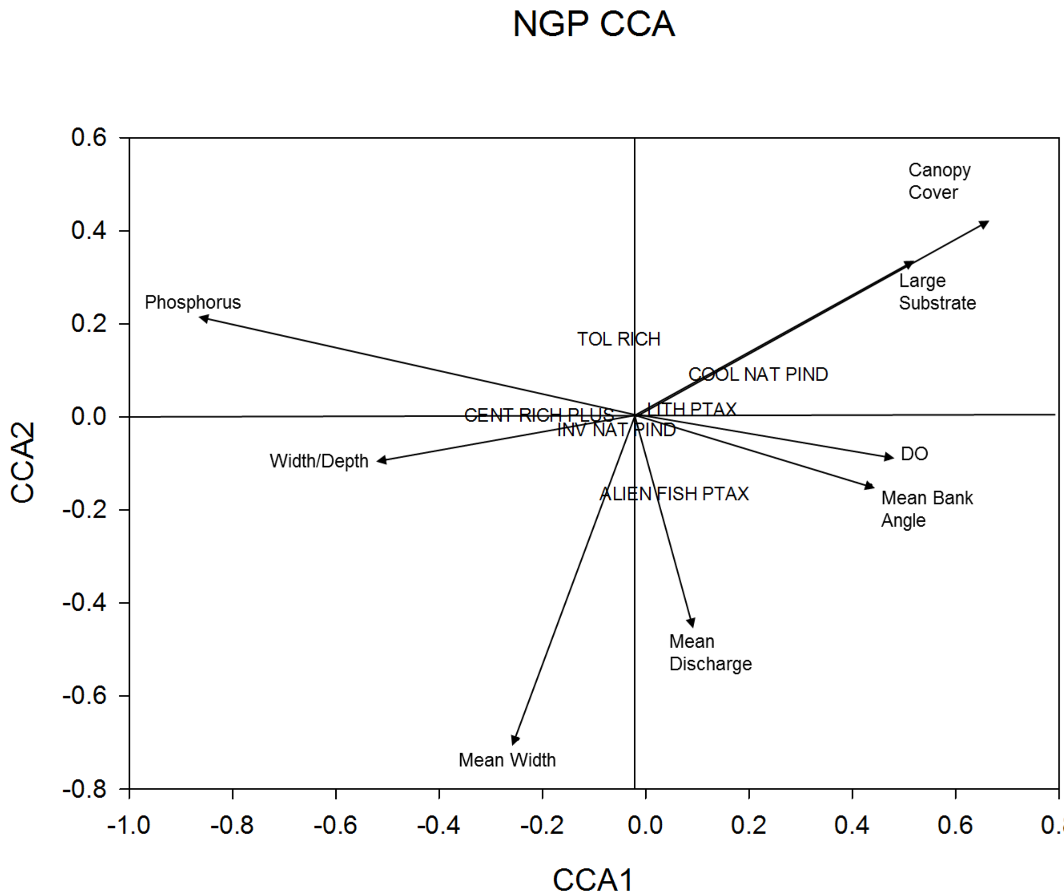

Fig. 3-2. Canonical Correspondence Analysis of the retained environmental variables with the NGP Index of Biotic Integrity (IBI) metrics plotted as species. Vectors are plotted with the associated habitat variable and the IBI metrics.

.

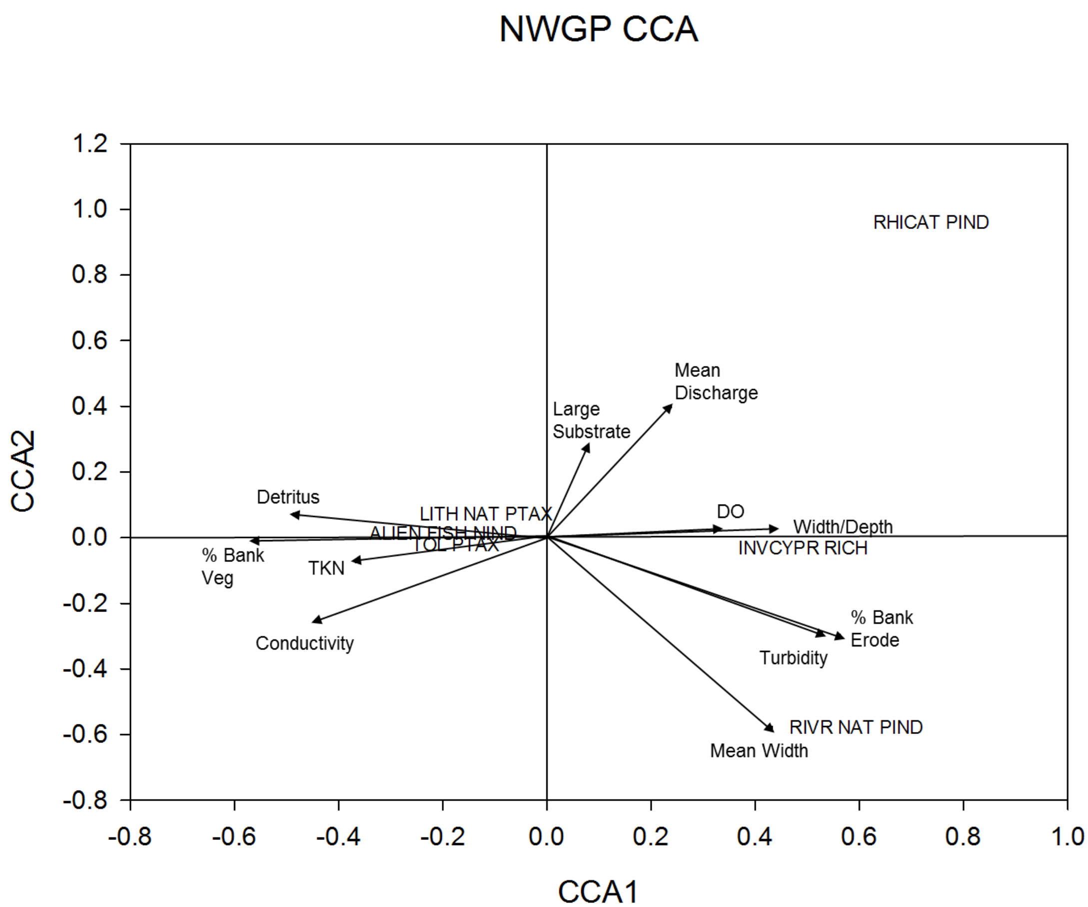

Fig. 3-3. Canonical Correspondence Analysis of the retained environmental variables with the NWGP Index of Biotic Integrity (IBI) metrics plotted as species. Vectors are plotted with the associated habitat variable and the IBI metric.

CHAPTER 4. SUMMARY

The first objective of this thesis was to develop a fish Index of Biotic Integrity (IBI) for the Northwestern Great Plains (NWGP) ecoregion. This was completed following sequential metric filtering (Whittier et al. 2007b) to produce a useable tool kit to assess stream fish assemblages in the NWGP ecoregion. . The second objective was viewed as an extension of current IBIs, the “what next” step after developing the fish IBI. Here, the relationships between environmental variables and fish assemblages were assessed by first determining which variables were correlated to IBI scores and then were assessed using the multivariate Canonical Correspondence analysis.

The IBI metric selection for the NWGP employed a chronological filtering process using range, signal-to-noise ratios, responsiveness to disturbance, and redundancy tests until there was one metric remaining in each class. Six metrics form six different metric classes passed the screening process from the initial 219 candidate metric pool in the NWGP ecoregion. These metrics represented both positive and negative interactions. The positive indicator metrics were Cyprinid Invertivore Species Richness, Proportion of Individuals that are Native Large River Migrants, Proportion of Individuals that are Longnose Dace (LOD) (Rhinichthys cataractae), and Proportion of All Species that are Lithophilic Spawners. The negative indicators of condition were the Proportion of All Species that are Tolerant and the Abundance of Alien Fish.

Seventy-five percent of the least disturbed sites scored between 60 and 85 out of 100 in the fish IBI, whereas over eighty percent of the most disturbed sites scored between 10 and 45. There was a significant statistical difference between scores of least disturbed sites and most disturbed sites ($F1,21 = 27.21$, $P < 0.00$) with least disturbed sites scoring 50% higher on average than most disturbed sites ($\bar{x}$=62.22 ± 5.06; $\bar{x}$=31.36 ± 2.77) (Fig. 3-3). The lowest average IBI scores were in the Missouri Plateau ($\bar{x}$=33.52 ± 7.33), Moreau Prairie ($\bar{x}$=40.83 ± 3.46), River Breaks ($\bar{x}$=35.29 ± 1.97), and Subhumid Pierre Shale Plains ($\bar{x}$=30.83 ± 3.49) ecoregions (Fig. 3-4). Compared to the highest scoring ecoregions the Keya Paha Table Lands ($\bar{x}$=56.31 ± 4.57), Sage Brush Steppe ($\bar{x}$=60.33 ± 4.99), and the White River Bad Lands ($\bar{x}$=67.22 ± 4.16). The NWGP IBI provides a tool for monitoring water quality in western South Dakota and a baseline of biotic condition in this ecoregion.

The second objective was to assess the relationship between environmental variables and assemblage structure. This chapter focuses on the question of why a site was scored what it was. The results of this work should allow to forecast potential fish habitat related stressors that affect assemblage distributions and either prevent degradation to critical fish community habitats or identify actionable habitat features for mitigation.

Two metrics (Proportion of All Species that are Tolerant and Abundance of Alien Fish) were influenced slightly by increases in conductivity. These metrics increased with decreases in dissolved oxygen and width to depth ratio. Cyprinid Invertivore Species Richness, Proportion of Individuals that are Native Large River Migrants, and Proportion of Individuals that are LOD increase with increases in dissolved oxygen and mean stream width by depth. The Proportion of Individuals that are LOD was strongly influenced by increases in mean discharge and the subsequent increase in substrate size. The Proportion of Individuals that are Native Large River Migrants increased predominantly with mean stream width, but appeared to be influenced by increases in turbidity and the amount of the bank that is eroded.

Increases in nitrogen and phosphorus can result in eutrophication, losses of diversity and decreases in dissolved oxygen, which can lead to fish kills (Carpenter et al. 1998). Increases in intensive agriculture activity has been shown to increase stream nitrogen and phosphorus (Bennett et al. 2001, Bernot et al. 2006). Intensification of agriculture land use can also effect the channel morphology, Gucker et al. (2009) found that streams in areas with increased agriculture activity were shallower and more homogenous. These habitat variables (phosphorus, DO, and width to depth) are the leading factors describing negative metrics in the NGP, and are could be mitigated at a single site, but can be complex to mitigate watershed-wide.   Riparian buffers and fences could be implemented to reduce the effects of sedimentation, grazing cattle, and chemical runoff. Borin et al. (2005) found that buffer strips decrease the amount of phosphorus that enters streams, specifically the sediment bound phosphorus, where buffer strips would also lower the amount of sediment entering the streams.

Reach level restoration efforts are vital to aquatic life. However, each segment of stream is not an isolated water body but the results of conditions upstream (Poff 1997; Marsh-Matthews and Matthews 2000; Jackson et al. 2001; Sindt et al. 2012). Without basin wide best management practices, fish that drive IBI scores will not have the ability to move or survive at specific points. What we have identified here are those elements that are essential to the assemblages that comprise each metric. This thesis is intended to serve as a tool to assess anthropogenic disturbance in the NWGP ecoregion and to assess the environmental variables driving site condition in the two largest ecoregions in South Dakota.

Literature Cited

Bennett, E. M. , S. R. Carpenter, and N. F. Caraco. 2001. Human Impact on Erodable Phosphorus and Eutrophication: A Global Perspective: Increasing accumulation of phosphorus in soil threatens rivers, lakes, and coastal oceans with eutrophication. BioScience 51 (3): 227-234.

Bernot, M. J., J. L. Tank, T. V. Royer, and M. B. David. 2006. Nutrient uptake in streams draining agriculture catchment of the midwest United States. Freshwater Biology. 51:499-509

Borin, M., M. Vianello, F. Morari, and G. Zanin. 2005. Effectiveness of buffer strips in removing pollutants in runoff from a cultivated field in North-East Italy. Agriculture, Ecosystems and Environment 105:101–114

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface water with phosphorus and nitrogen. Ecological Applications 8(3):559-568

Gucker, B., I. G. Boechat, and A. Giani. 2009. Impacts of agriculture land use on ecosystem structure and whole-stream metabolism of tropical Cerrado streams. Freshwater Biology. 54:2069-2085

Jackson, D. A., P. R. Peres-Neto, and J. D. Olden. 2001. What controls who is where in freshwater fish communities-the roles of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58:157–170.

Marsh-Matthews, E., and W. J. Matthews. 2000. Geographic, terrestrial and aquatic factors: which most influence the structure of stream fish assemblages in the midwestern United States? Ecology of Freshwater Fish 9:9–21.

Poff, N. L., J. D. Allen, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47(11):769-784.

Sindt, A.R., M.C. Quist, and C.L. Pierce. 2012. Habitat associations of Fish species of greatest conservation need at multiple spatial scales in wadeable Iowa streams. North American Journal of Fisheries Management 32: 1046-1061.